

NAVIGATING NUANCE IN NATIVE BEE RESPONSES TO GRASSLAND RESTORATION  
MANAGEMENT: A MULTI-ECOREGIONAL APPROACH IN THE GREAT PLAINS

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The following faculty members have examined the final copy of this thesis for form and content and recommend that it be accepted in partial fulfillment of the requirement for the degree of Master of Science with a major in Biological Sciences.

Mary Jameson, Committee Chair

Gregory Houseman, Committee Member

Doug English, Committee Member

“The idea of wilderness needs no defense, it just needs defenders.”—Edward Abbey

“The conservation of natural resources is the fundamental problem. Unless we solve that problem it will avail us little to solve all others.”—Theodore Roosevelt

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## **CRP Plants, Insects & Birds**

## ABSTRACT

Habitat fragmentation due to agricultural intensification leads to losses of biodiversity and ecosystem services such as pollination. Wild bee declines pose a serious threat to pollination stability and are expected to be most severe in agricultural landscapes, providing the impetus for grassland restoration efforts like the Conservation Reserve Program (CRP) to establish pollinator habitat via forb-enhanced plantings. One form of management not applied to CRP forb-enhanced plantings is grazing. Given historical adaptation of Great Plains ecosystems to large grazers such as bison, it is possible that grazing-induced benefits to pollinators and forb communities have been overlooked. Few studies have examined the effects of restoration management on grassland native bees. Specifically, there is need to a) assess the effects of restoration plantings on native bees throughout Great Plains ecoregions, and b) examine how associated management practices, such as grazing, affect bee communities. This study is the first of its kind to assess bee and forb responses to grazing and restoration plantings across multiple ecoregions of the Great Plains.

We surveyed native bee and forb communities on 108 CRP fields throughout the sand, short-, mixed-, and tall-grass prairie ecoregions of Kansas. CRP fields were either a) forb-enhanced or primarily grass-planted and b) not grazed or grazed at low-intensity. Overall, we found that floral cover was the most important predictor of bee responses. Forb-enhanced restorations did not differ in floral cover or richness, but still had positive effects on native bees. Bee community responses to grazing were mostly positive with one exception: bee diversity decreased on mixed-grass, grazed fields. Additionally, bee communities responded to grazing only after grazing had ceased. Our findings provide novel insight into bee community responses to land management on restored grasslands in a predominantly agricultural Great Plains landscape.

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## INTRODUCTION

Habitat fragmentation and loss, as a result of widespread conversion of natural areas for human use, threatens the ecological functioning of many global biomes (Hoekstra et al. 2005, Fahrig 2003). In the Great Plains region, which is characterized primarily by temperate grasslands, conversion of habitat to cropland has led to losses of as much as 99% of native prairies (Samson and Knopf 1994). Subsequent consequences for biodiversity due to such extreme habitat loss are associated with declining ecosystem services (Dobson et al. 2006, Tilman 1999, Potts et al. 2010), and one keystone service currently at risk is animal-mediated pollination (Kearns et al. 1998, NRC 2007). Insects comprise a majority of pollinators and are essential to the reproductive success of a majority of flowering plants (Klein et al. 2007, Ollerton et al. 2011). Bees are the most effective pollinators of many native plant and crop species and are therefore indispensable to both agricultural and natural ecosystems (NRC 2007, Ollerton et al. 2011, Kearns and Inouye 1997); however, the impacts of habitat fragmentation and loss on bee populations are inversely related to remaining natural habitat (Winfree et al. 2009). As natural habitat is converted to cropland, floral resources are removed and nesting habitat is destroyed, typically resulting in decreased wild bee abundance and diversity. Thus, in the United States, bees are expected to be at greatest risk of decline in the highly modified landscapes of the Great Plains (Koh et al. 2016).

Although the production of bee-pollinated crops such as almonds and blueberries traditionally rely on managed, non-native honeybees (*Apis mellifera*), recent colony declines and increased demands for crop pollination services (Ward et al. 2010) are the impetus for alternative solutions (Drummond et al. 2017). While over 4,000 native North American bee species with diverse life histories and habitats are capable of buffering against honeybee losses (Winfree et al.

2007, Rader et al. 2012, Garibaldi et al. 2011), wild bees require habitat and resources beyond the large-scale monocultures characteristic of US agriculture. At a minimum, we know that bees require diverse and abundant foraging resources (i.e. flowering plants) and nesting habitat. Because mass blooming crops can provide only temporary forage for wild bees, floral resources with season-long blooms are necessary to support diverse bee communities with temporal complementarity (Holzschuh et al. 2012, Kremen et al. 2007). Additionally, the stability of crop pollination services provided by native bees is directly related to the amount of surrounding natural habitat (Kremen et al. 2004, Garibaldi et al. 2011, Ricketts et al. 2008). Natural and semi-natural areas interspersed throughout otherwise simplified agricultural landscapes can provide diverse and abundant floral communities and nesting resources (Hopwood 2008, Ricketts et al. 2008, Williams et al. 2010) while simultaneously buffering against adverse effects of habitat fragmentation (Winfree et al. 2009) and contributing to the resiliency of pollination services in agroecosystems (Kremen et al. 2004).

Native bees vary substantially in nesting biology, floral preferences, and foraging ranges, which are directly related to bee body size (Greenleaf et al. 2007). Thus, effective pollinator habitat must provide not only sufficient floral and nesting resources at the patch-level (Williams and Kremen 2007, Roulston and Goodell 2011), but must occur with enough regularity throughout an agricultural landscape to provide spatial complementarity of resources and facilitate landscape connectivity (Rao and Strange 2012, Tscharntke et al. 2005, Menz et al. 2011). For this reason, large-scale efforts to conserve and establish natural and semi-natural habitat, often in the form of restoration plantings, may be the most valuable tool for the conservation of both native bee communities and the stability of pollination services in agroecosystems by connecting habitat and promoting landscape heterogeneity. One such

example of conservation in the United States is the Conservation Reserve Program (CRP) which is one of the largest grassland habitat restoration efforts in the Great Plains. The 22 million acres of land currently set aside by the CRP (USDA 2019) thus contribute extensively to agroecosystem restoration efforts.

Established in 1985 by the United States Department of Agriculture (USDA), the CRP provides annual rental payments as incentive for landowners to voluntarily retire marginal and environmentally sensitive cropland for 10-15 years and implement approved restoration types, collectively referred to as ‘Conservation Practices’ (CP), with specific, targeted conservation objectives (USDA 2013). Originally targeting soil and water-quality improvement, wildlife and pollinator habitat restoration have since been incorporated in the overarching goals of the CRP.

Upon enrollment, CRP-participating landowners follow a specific set of guidelines that stipulate allowable initial plantings. Theoretically, initial plantings aim to establish certain plant species on restorations, and management regimes ensure these grassland plant communities remain in an early successional stage by using disturbance to prevent woody encroachment (Twidwell et al. 2013, Gruchy et al. 2006). Evaluations of potential CRP benefits to pollinators have typically focused on the inclusion of forb species in initial CP plantings (Vaughan and Skinner 2015, Haley 2018). Additionally, at least two Conservation Practices, CP-25 and CP-42, explicitly focus on the establishment of pollinator habitat as a primary objective (USDA 2015*a,b*). Given overwhelming evidence that floral resources (i.e. density, cover, richness, diversity) are positively related to wild bee abundance, richness, and diversity (Morandin and Kremen 2013, Williams et al. 2015, Carvalheiro et al. 2011, Hoehn et al. 2008), forb-enhanced restoration plantings are expected to, at the very least, result in greater floral resource availability to pollinators compared with plantings that do not incorporate forb species. However, minimal

data currently exist to verify that forb-enhanced CRP plantings result in successfully established floral communities that in fact support diverse and abundant pollinator communities, thus serving as effective pollinator habitat.

CRP enrollment guidelines also specify permitted mid-contract management options to mimic natural disturbances (USDA 2013). While the plant species included in initial restoration plantings are CP-specific, options for mid-contract management practices are fewer and typically include disking, spraying, interseeding, haying and, less commonly, grazing. Great Plains grasslands were historically dominated by grazing bison (Freese et al. 2007, Sanderson et al. 2008), and prairie plant species are generally adapted to grazing disturbances (Augustine and McNaughton 1998). However, cattle grazing is currently disincentivized on CRP lands. While some Conservation Practices permit grazing at a payment reduction, others, including pollinator habitat CPs, prohibit grazing altogether (USDA 2013).

Overall, the effect of grazing on bee communities is poorly understood. Some studies have linked high-intensity grazing with decreased bee and pollinator richness (Hatfield and LeBuhn 2007, Sjödin 2007, Le Féon et al. 2010) and short-term, low-intensity grazing with unaltered or increased bee abundance (Steffan-Dewenter & Leschke 2003, Carvell 2002). A majority of studies have examined the indirect effects of grazing on pollinators via changes in plant communities (Stoner and Joern 2004, Hayes and Holl 2003, Berg et al. 2019), though a few have also documented direct effects of grazing on bee nests and soil-mediated effects on nest habitat suitability (Sugden 1985, Kimoto et al. 2012). Overall, the impacts of grazing on native bees seem to vary with the type of grazer (Hatfield and LeBuhn 2007), intensity of grazing (Kimoto et al. 2012, Carvell 2002), timing (Sjödin 2007), local habitat (Minckley 2014), and historic adaptation of location to grazers (DeBano et al. 2016). Thus far, however, no systematic

research efforts that we know of have documented the effects of grazing on bee communities: a) between restoration grasslands differing in initial plantings, or b) at a broad geographical scale encompassing multiple grassland ecoregions.

This study aims to assess bee communities on CRP grasslands and identify the primary factors driving native bee abundance, genus-richness, diversity, and community composition across four prairie ecoregions and between two restoration types (CP) and a grazing treatment. We surveyed bee and forb communities on 108 CRP fields across 650 km of the Great Plains region and our study area included short-grass, sand, mixed-grass, and tall-grass prairie ecoregions spanning an annual precipitation gradient of 63.5 cm (45.7–109.2 cm). CRP fields in this study were split between two Conservation Practices: CP-2, a grass-planting restoration targeting the improvement of soil and water quality, and CP-25, a forb-enhanced restoration for the conservation and establishment of wildlife and pollinator habitat. Additionally, we implemented a grazing treatment and CRP fields were either grazed by cattle at a medium-to-low-intensity stocking rate or not grazed (ungrazed).

Our primary goal is to identify key factors driving bee community responses to grassland restoration practices in Great Plains agroecosystems. Specifically, our research: 1) assessed differences in bee communities between two restoration types (forb-enhanced vs. grass plantings), 2) evaluated bee community responses associated with grazing-induced changes to restored grasslands, and 3) examined differences in native bee community responses to grazing and restoration type across four grassland ecoregions. Overall, we expected the effects of grazing and restoration type on bee communities would be detectable in both years of the study. We predicted that, compared with grass plantings, forb-enhanced restorations would support more diverse and abundant bee communities through greater floral resource availability. Because

Great Plains grasslands are historically adapted to the presence of large grazers, primarily bison (Knapp et al. 1999), we expected low-intensity, short-term cattle grazing would positively affect forb, and subsequently bee, abundance and diversity. Finally, given expected abiotic and biotic differences between ecoregions, including inherent differences in bee communities, we predicted that the extent of forb and bee responses to grazing and restoration type would vary between sand, short-, mixed-, and tall-grass prairie ecoregions.

## METHODS

### *Ethics statement*

Field work was conducted with permission of private landowners and in concert with USDA and FSA. This research did not include any endangered or protected Hymenoptera species.

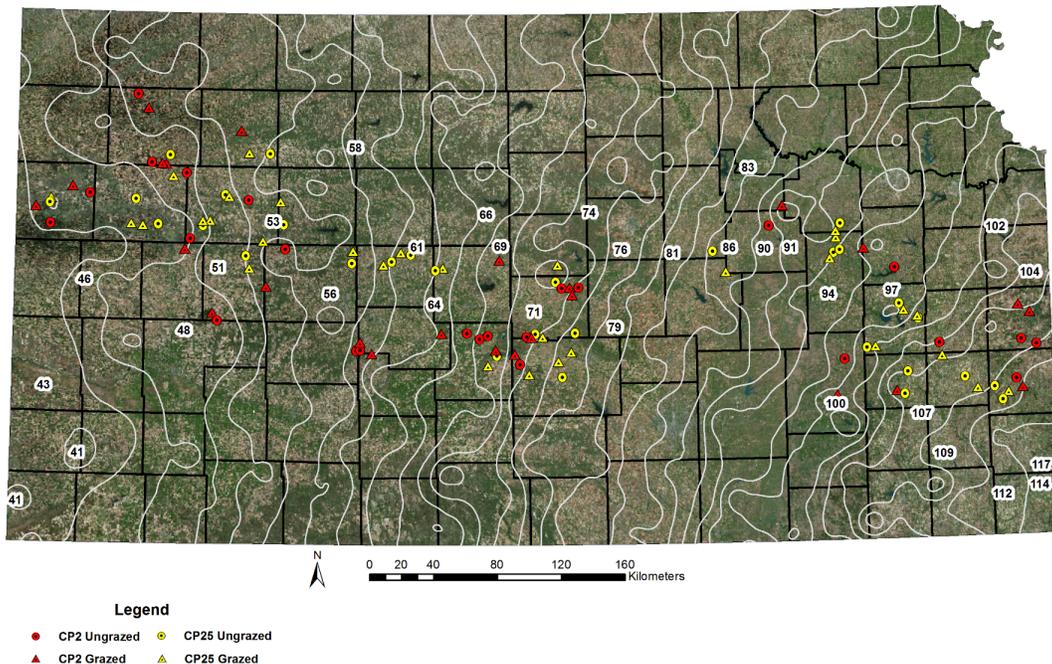
### *Site selection and plot set-up*

From mid-May to late July in 2018 and 2019 we surveyed forb and bee communities on Conservation Reserve Program (CRP) land in sand, short-, mixed-, and tall-grass prairies of Kansas. The study area spanned 650 km from east to west and a precipitation gradient of 63.5 cm (from 45.7–109.2 cm annually based on 30-year averages). We selected 108 CRP-enrolled fields using a randomized CRP landowner call list generated from the United States Department of Agriculture (USDA) database. In order to be included in the study, all fields were a) enrolled in the CRP prior to 2012 (and with contracts not expiring before 2019), b)  $\geq 35$  acres, and c)  $>1$  km from the nearest site.

For individual CPR-enrolled fields, the Natural Resource Conservation Service (NRCS) assigns specific Conservation Practices (CP) under which CRP landowners can enroll and manage their land. Each CP, differentiated by arbitrarily-assigned numbers (i.e. CP-2), specifies targeted conservation goals, the identity and quantity of plant species to be included in initial seeding mixes, and permissible management practices (USDA 2013). The CRP fields in this study were split between the two most common conservation practices implemented in the state of Kansas (USDA 2019): CP-2 (n=52) and CP-25 (n=56) (Fig. 1). Given that CP-2 was one of the original Conservation Practices of the CRP (via the 1985 Farm Bill), and CP-25 was not

implemented until more than 10 years later (Nicole Welborn, Kansas FSA *personal communication*; Oct.16, 2019), CP-2 plots were, on average, 14 years older than forb-enhanced CP-25 fields.

CP-2 is designated for the establishment of permanent and native grasses. The primary conservation goals of CP-2 include wind and water control, water quality enhancement, and the creation of habitat for grassland and game birds (NRCS 2015a).



**FIG. 1** Area of study throughout Kansas with 108 CRP fields. CP-2 fields are given in red, with CP-25 in yellow. Grazed fields are represented by triangles and ungrazed fields are represented by circles. White lines show annual precipitation isoclines, as determined by 30-year averages. (USDA NRCS–National Geospatial Center of Excellence 2012). Image by Jackie Baum.

Landowners with CP-2-enrolled land are required to plant at least two native grass species and are incentivized to plant up to two forb species, although the inclusion of forbs is not required (NRCS 2012a).

CP-25 focuses primarily on the rehabilitation of rare and declining habitats, specifically aiming to the reduce soil erosion, restore wildlife habitat, and create pollinator habitat through

the establishment of forb communities (USDA 2015*b*). In Kansas, the rare and declining habitats identified for restoration are short-grass, sand, sand-sage, mixed-grass, and tall-grass prairies. CP-25 landowners are required to plant five native grass species and a minimum of four (short-grass) or ten (tall-grass) native forb species. Planting guidelines for CP-25 allow only the inclusion of plant species ecotypical to a respective rare and declining habitat (NRCS 2012*b*). CRP fields included in this study were distributed across all Kansas NRCS habitat types except sand-sage prairie.

Given the 650 km longitudinal gradient encompassed by our study, we included NRCS habitat types (herein referred to as ‘ecoregions’) in our analyses in order to account for the environmental and geographical variation across study fields. CP-25 planting guidelines characterize the distribution of prairie habitats at the county level (NRCS 2012*b*). Because some counties were associated with two habitat types, we cross-referenced NRCS habitats with the respective NRCS Common Resource Areas (CRA’s; NRCS 2004) and EPA IV ecoregions (Appendices A and B, Chapman et al. 2001) using GPS coordinates for all CRP fields. Of 108, we were unable to reach a uniform consensus for 11 CRP fields; for these we deferred to the original NRCS designations, as stipulated by CP-25 planting guidelines (Appendix C). For this study, sand prairie and short-, mixed-, and tall-grass prairies are collectively referred to as ecoregions (Appendix C).

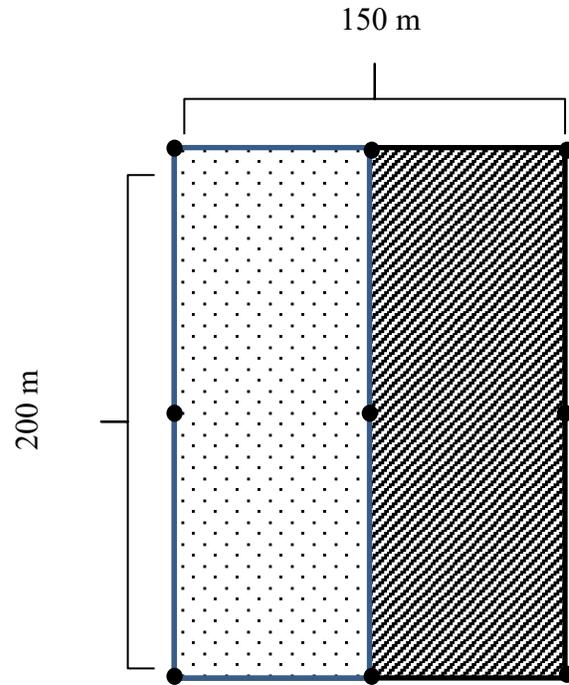
Although CP-2 landowners are allowed to graze cattle as a mid-contract management option at a slight payment reduction, CP-25 landowners are explicitly prohibited from grazing. However, in order to assess the effects of grazing on forb-enhanced restorations in this study, 29 CP-25 landowners were given special permission to graze by the Farm Service Agency (FSA). These fields, as well as 24 enrolled under CP-2, were grazed by cattle for two years (2017-2018);

fields were not grazed in 2019. Field-specific stocking rates were determined by the NRCS for CRP fields in this study and targeted 50% plant biomass reduction.

### *Bee collection and identification*

We established 150 x 200 m study plots in the center of each CRP field, within which we placed nine equally-spaced sampling points (Fig. 2). Bee and plant communities on each plot were sampled twice between May and July in 2018 and 2019. To sample CRP bee communities, we hand-netted all floral visitors from flowering forbs for 30 minutes per plot visitation. Unlike hand-netting, other common bee sampling techniques such as vane and pan traps are shown to attract only a subset of bee taxa, and sampled abundance and diversity can be inversely variable with flower cover (Mayer 2005, Baum and Wallen 2011, Wilson et al. 2008, Cane et al. 2000). Hand-netting allows collectors to capture taxa not typically attracted to pan traps, eliminates competition with the surrounding floral resources, and provides more complete samples of bee communities (Minckley 2014, Roulston et al. 2007).

Study plots were split lengthwise into two 75 x 100 m hand-netting transects (Fig. 2). Two technicians simultaneously walked the length of each transect for 15 minutes, visually locating and hand-netting flowering forb patches within respective transect boundaries. To avoid biases towards larger species, all forbs visited were hand-netted regardless of visual confirmation of bee presence. For both years of the study, we conducted two sampling events on each plot (4-5 weeks apart) from May 23 – July 25 in 2018 and May 20 – July 26 in 2019. During each visit, two of three possible technicians hand-netted a transect for fifteen minutes in order to avoid



**FIG. 2** Study plots utilized in this study measured 200 x 150 m. Individual bee hand-netting transects are indicated by the dotted and striped areas within the study plot (200 x 75 m). Quadrats (1 x 1 m) were placed at each of the nine sample points (black dots) for conducting plant community surveys and quantifying mean flowering forb percent cover and flowering forb species richness.

collector bias. Together, the two fifteen-minute sampling periods totaled 30 minutes of netting per plot per visit, with each site hand-netted for 60 minutes per year. Hand-netting was completed between 08:00 and 18:35 h, with 94.5% of all samples collected during the optimal hours of bee activity, between 9:00 and 18:00 h (following Geroff et al. 2014). Hand-netted samples were frozen within the same day of collection. In the laboratory, samples were sorted to and bees were separated from any additional flower-visiting insects vicariously collected in the process of hand-netting flowering forbs. All bees were curated and identified to genus by Alex Morphew (Wichita State University) using genera-level keys (Michener et al. 1994, Michael Arduser, *personal communication*, March 2019). Michael Arduser verified identifications and subsequently

identified all bees to the lowest possible taxonomic level for future analyses (ongoing). Upon completion of species-level identifications, voucher specimens will be deposited at Wichita State University Invertebrate Collections (Wichita, KS; WICHI), University of Colorado Museum (Boulder, CO; UCMC), and Snow Entomological Museum Collection at the University of Kansas (Lawrence, KS; SEMC).

#### *Flowering forb data collection*

For each plot, plant and bee communities were sampled on the same day. At each of the nine sample points within a study plot (Fig. 2), we identified all entomophilous flowering forb species within a 1 m<sup>2</sup> quadrat. In addition to species identity, we estimated the percent cover occupied by each forb species. For a majority of sampling events, the same individual performed all plant identifications and estimations of percent cover in all years of the study and three plant technicians were each primarily responsible for flowering-forb data collection on 36 plots (Alex Morphew in western Kansas, Esben Kjaer in central Kansas, and D. Fraser Watson in eastern Kansas; all Wichita State University).

#### *Univariate statistical analyses*

We pooled all hand-netted samples across collectors and plots and calculated bee abundance, genera richness, and diversity for 2018 and 2019. Bee diversity was measured as effective genera number ( $e^H$ ; Jost 2006), herein referred to as ‘diversity’. For 2018 and 2019 plant communities, we determined the quadrat-level species richness of entomophilous forbs in flower at the time of sampling. We pooled floral richness across all quadrats and both sampling events. To determine the abundance of entomophilous floral resources, we calculated the average

percent cover of flowering forbs at the quadrat-level, pooled across both sampling events. Together, floral richness and floral abundance were used to estimate effects of floral resource availability in analyses.

To evaluate relationships between univariate metrics of bee community response (abundance, richness, and diversity) and: a) restoration planting type (CP-2 and CP-25), b) grazing treatments (grazed and ungrazed), and c) CP-grazing, CP-ecoregion, and grazing-ecoregion interactions, we utilized a multimodel inference and model averaging approach (Burnham and Anderson 2002; Cade et al. 2015). This method minimizes the influence of unimportant predictors (Arnold 2010) and type I error typically associated with testing complex generalized linear models (Burnham and Anderson 2002). We assessed the distribution of each response variable and applied data transformations when necessary to achieve normality. Bee abundance was log-transformed to fit a Gaussian distribution and a Gamma-family general linear model (GLM) was utilized for assessing diversity. Bee genera richness counts were overdispersed and thus required a negative binomial GLM (Gardner et al. 1995). Prior to constructing candidate model sets, we tested for multicollinearity of predictor variables within and between initial candidate sets because the retention of multicollinear variables within the same models can lead to distorted model estimations (Cade 2015; Dormann et al. 2013). For each year of the study, we calculated Spearman's correlation coefficients to determine if floral cover and floral species richness were collinear, utilizing a threshold of 0.5 for inclusion of explanatory variables in the same models. To compare floral resource covariates with CP and grazing predictor variables we conducted non-parametric Wilcoxon rank-sum tests. To test for correlations between floral resources and ecoregion, non-parametric Kruskal-Wallis rank-sum

tests and parametric Analyses of Variance (ANOVA) were utilized when appropriate. For all tests of collinearity with our categorical predictors, we established a threshold of  $P < 0.10$ .

When multicollinearity was detected, we modeled collinear predictor variables separately against bee abundance, genus richness, and diversity. We compared adjusted Akaike Information Criterion scores (AICc), for abundance and diversity, and Quasi-AICc (QAICc), for genera richness (Burnham and Anderson 2002), between correlated predictor pairs to determine which variable better estimated each bee response metric, based on a threshold of AICc and QAICc differences ( $\Delta\text{AICc}$ ,  $\Delta\text{QAICc}$ )  $< 4$ . When floral resource variables out-competed categorical treatment variables, we constructed additional model sets for preliminary model comparison, based on the structure of the original treatment-effects model set. To create these ‘modified’ models, we substituted floral resource predictor variables for the treatments with which they were collinear and eliminated subsequent models in which collinear variables co-occurred.

Preliminary candidate model sets for bee abundance, richness, and diversity were compared separately to ascertain the most informative parameter combinations. The preliminary model sets for our three metrics of bee community response in both 2018 and 2019 were: a) floral resources (cover and species richness), b) ‘treatment’ (grazing, ecoregion, CP and their interactions), and c) modified floral-treatment substitutions in the event of collinearity. Each model set also contained the null model for the associated bee response metric in order to safeguard against Type I errors (Forstmeier and Schielzeth 2011). We examined the residual errors and variance for each global model in preliminary model sets to ensure that all general linear models met assumptions of normally-distributed residuals and homoskedastic variance (Breslow 1996). To ensure inclusion of all categorical treatment variables of interest, original model sets containing the full suite of treatment parameters were retained in preliminary model

comparisons across all analyses, regardless of whether floral resource variables better estimated bee response variables.

Within each preliminary model set, we compared AICc values across models to select the best-estimating models. Combinations of predictor variables comprising models with differences in AICc ( $\Delta\text{AICc}$ )  $< 4$  were retained for inclusion in the final combined candidate set. For each metric of bee response and year, we established a final, combined model set containing all original models with  $\Delta\text{AICc} < 4$  from preliminary comparisons in addition to models with predictor variable combinations merged across preliminary model sets, except in instances of multicollinearity. This ‘final’ candidate set of models was then subject to hierarchical multimodel inference. Model weights were calculated for every model in the final candidate set, as well as evidence ratios (Burnham and Anderson 2002). The model with the lowest AICc-value ( $\Delta\text{AICc}=0$ ) was considered the best model, and models with  $\Delta\text{AICc} < 4$  were considered to be competitive and retained for multimodel inference. For this top model set, we followed a hierarchical decision-making process using a series of “warning signals” (Appendix D, Leroux 2019) to identify models with uninformative parameters. Thus, we retained only informative models when averaging parameter coefficients and confidence intervals to obtain parameter estimates (Anderson 2010, Leroux 2019). In order to achieve comparable coefficient estimates across models, all parameter estimates were standardized based on the partial standard deviations for their variables (Cade 2015).

All analyses were implemented in the statistical software R v.3.5.2 (R Core Team 2018). The package MuMIn v. 1.42.1 (Bartón 2018) was used for multimodel inference and model averaging.

### *Multivariate statistical analyses*

In order to compare site-level bee community composition between CP, grazing, ecoregion, and subsequent two-way interactions, we conducted repeated-measures permutational analysis of variance (PERMANOVA) using Bray-Curtis dissimilarity and relative bee abundance data from 2018 and 2019. Prior to conducting PERMANOVA analyses, we removed all plots on which no bees were collected (Table 1), as well as all singletons in the form of: a) bee genera represented by only one individual, b) genera that occurred on only one plot, and c) plots on which only one genus was collected. Singletons were removed to minimize large differences in bee community dispersion, which can lead to erroneous p-values. We tested for homogeneity of group dispersions (PERMDISP function in PRIMER-E version 6; Clarke and Gorley 2006) to assess whether bee community composition was significantly more variable between CP, grazing, ecoregion, and subsequent pairwise interactions. When necessary, we assessed compositional differences of bee communities among ecoregions using pairwise comparisons. Additionally, indicator genera analyses were used to identify specific bee genera significantly associated with treatment effects (R package “indicspecies”, De Caceres and Legendre 2009). All PERMANOVAs were conducted in PRIMER-E version 6 (Clarke and Gorley 2006). In order to visualize significant bee community compositional differences, we used nonmetric multidimensional scaling (NMDS) and Bray-Curtis dissimilarity to produce two ordination plots (R package “ggplot2” version 5, Wickham 2016). Vectors for floral resource measures (floral richness and floral cover) were included in ordination plots to visualize associations with NMDS axes, but we do not report p-values due to high stress values.

## RESULTS

### *Native bee genera*

In 2018 and 2019, we collected a total of 10,970 bees on 120 entomophilous forb species. Bees were collected during 432 sampling events on 108 Kansas CRP fields within four grassland ecoregions over two field seasons (2018 and 2019). All bee identifications were resolved to genus level for a total of 40 bee genera (Appendix E) of the approximately 56 known to occur in Kansas (based on KSEM and AMNH databases accessed Sept. 2019). By abundance, approximately 86% of all bees were of the family Halictidae and belonged to four primarily semi-social genera: *Lasioglossum* (59.4%), *Augochlorella* (13.5%), *Agapostemon* (7.3%), and *Halictus* (4.7%). The semi-social nature of these bee genera accounts, in part, for their high abundance.

Of the genera collected, 20% were rare (1 or 2 specimens) and 7.5% were somewhat rare (3-10 specimens) (Appendix E). Rare genera included *Anthidium*, *Ashmeadiella*, *Dianthidium*, *Epeolus*, *Heriades*, *Xenoglossa*, and *Xeromelecta*. Of these, *Anthidium*, *Dianthidium*, *Ashmeadiella*, and *Heriades* are all members of the family Megachilidae and primarily nest in pre-existing cavities in wood or the voids of pithy stems (Fabre 1914, Michener 2007). *Xenoglossa* is one of two specialist genera that exclusively forage on *Cucurbita* (squash) flowers (Hurd et al. 1971), thus accounting for its low abundance on restored grasslands. *Epeolus* typically parasitizes the nests of *Colletes* (Onuferko 2017), and *Xeromelecta* is cleptoparasitic on the bee of the genus *Anthophora* (Linsley and MacSwain 1942, Porter 1951). These genera are found in low abundances in studies documenting diversity in short-grass prairies (Arathi et al. 2019) and tall-grass prairies (Haley 2018). Of note is the low abundance of bees in the genus *Eucera*, which are solitary, early- to mid-season bees that nest in clay or sand soils and

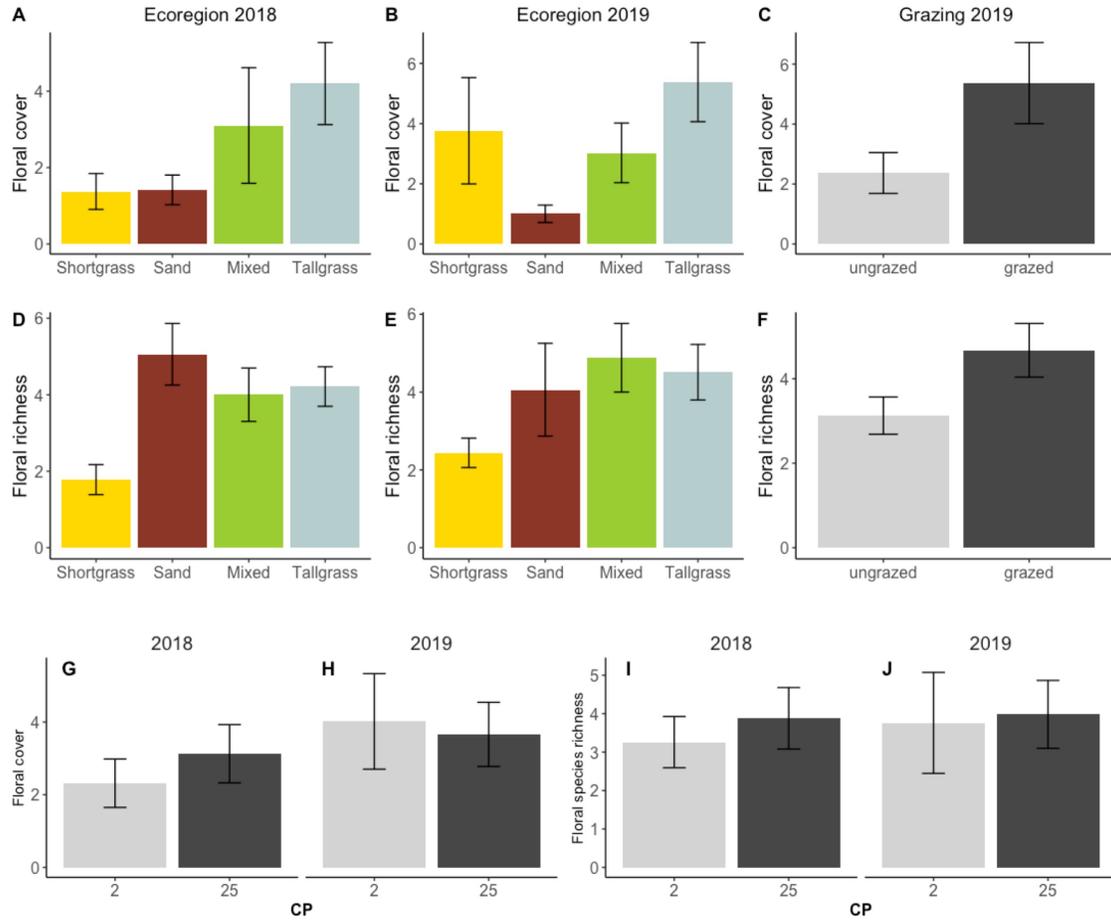
which were found in relatively high abundance in other studies assessing bee communities on CRP grasslands (Arathi et al. 2019, Haley 2018). Of the 108 fields surveyed, no bees were found on two plots in 2018 and six plots in 2019. Of these, samples from one plot (CP-2, ungrazed) had no bees in both years (Table 1). Three zero-abundance plots were ungrazed/CP-2 in the sand prairie ecoregion; two plots were ungrazed/CP-25 in the tallgrass prairie ecoregion; and two were grazed/CP-2 in shortgrass and tallgrass prairies (Table 1). Of these zero abundance plots, we recorded no more than three entomophilous flowering forb species on any plot, and overall, mean floral cover did not exceed 1% (Table 1).

**Table 1** Year, ecoregion, restoration type (CP), grazing treatment, floral cover, and floral richness for plots on which no bees were collected. ‘Plot code’ represents the identity of plots, designated by: a) the letter indicating the plot-grouping associated with a specific plant identification technician (W, C, E for west, central, and east Kansas) and b) an arbitrarily assigned number between 1-36 (the number of plots in each grouping). On plot C30 (gray) no bees were collected in either year.

Year	Plot Code	CP	Grazing	Ecoregion	Mean % floral cover (quadrat)	Floral richness
2018	C30	2	ungrazed	Sand	0.28	1
	W23	2	grazed	Short-grass	0	0
	C29	2	ungrazed	Sand	0.99	2
2019	C30	2	ungrazed	Sand	0.06	1
	C35	2	ungrazed	Sand	0.58	3
	E26	2	grazed	Tall-grass	0	0
	E35	25	ungrazed	Tall-grass	0.21	2
	E36	25	ungrazed	Tall-grass	0.04	1

*Multimodel inference and model averaging*

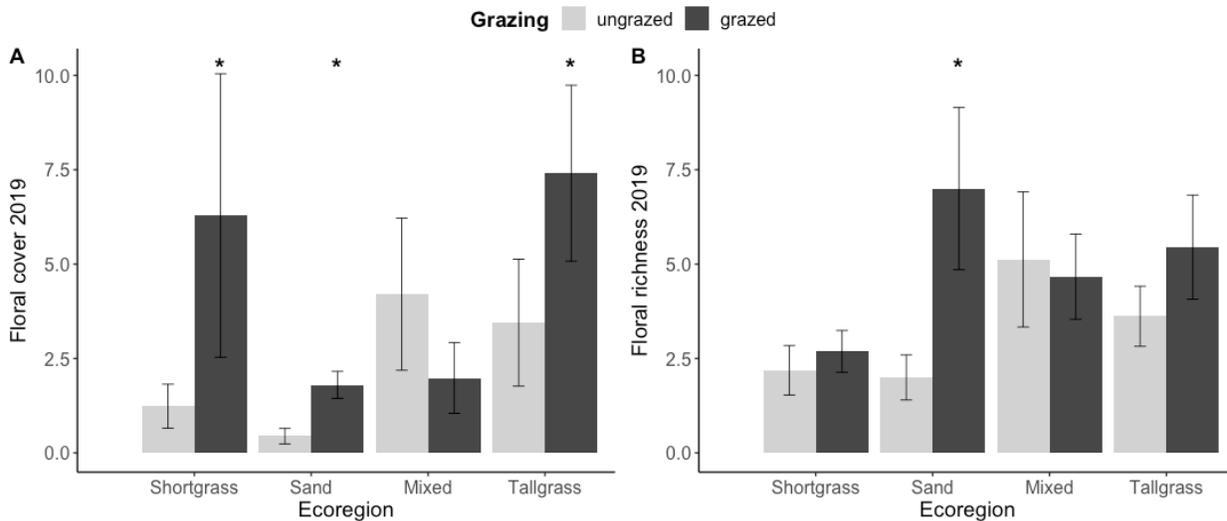
Both metrics of floral resource availability, floral cover and floral species richness, were collinear with ecoregion in 2018 and 2019 (Figs. 3 A, B, D, and E, Appendix F). Neither were correlated with restoration type (CP): neither floral cover nor floral species richness were



**FIG. 3** Mean floral cover (A, B, and C) and floral richness (D, E, and F) given by ecoregion, grazing, and year. Floral cover and floral richness were collinear with ecoregion (A, B, D, and E) in both 2018 (A and D) and 2019 (B and E). Floral cover and floral richness were collinear with grazing (C and F) in 2019. Neither floral cover (G and H) nor floral richness (I and J) were correlated with CP in either year. The height of the bars indicates means by group and year. Error bars represent 85% confidence intervals.

significantly associated with forb-enhanced plantings (CP-25) (Figs. 3 G-J). Additionally, floral cover and restoration type were not correlated with one another in year (Appendix F). Along with ecoregion, both floral cover and floral richness were collinear with grazing in 2019 (Figs. 3 C and F, Appendix F). Given collinearity between floral resources and both ecoregion and grazing, we conducted t-tests for all ecoregional pairwise interactions between grazed and ungrazed floral cover and floral species richness to identify interactional differences. In all ecoregions except mixed-grass prairie, floral cover was significantly higher on grazed fields than

ungrazed fields (Fig. 4 A,  $t_{15.73}=2.180$ ,  $p=0.045$  for short-grass;  $t_{9.87}=5.414$ ,  $p=0.0003$  for sand; and  $t_{34.93}=2.263$ ,  $p=0.030$  for tall-grass). Floral richness however, was only significantly different between sand prairie grazing treatments (Fig. 4 B,  $t_{6.92}=3.678$ ,  $p=0.008$ ), and grazed fields had, on average, more species of flowering forbs than did ungrazed fields.



**Fig. 4** Pairwise comparisons of floral cover (A) and floral richness (B) on grazed (dark gray) and ungrazed (light gray) plots across ecoregions in 2019. Bar heights represent means and error bars indicate 90% confidence limits. Asterisks (\*) designate significant differences ( $p < 0.10$ ) within ecoregions.

When floral resource variables were modeled against each measure of bee response and compared to collinear treatments, floral cover and species richness were overall better predictors of bee metrics than ecoregion or grazing ( $\Delta AICc < 4$ , Appendix G). There were, however, a few exceptions. Ecoregion was a more superior predictor of bee abundance and genus-richness than either floral resource metric in 2019. Furthermore, in 2019, separate models containing the term for grazing and floral richness comparably estimated bee richness ( $\Delta AICc < 4$ ), but bee abundance was better estimated by the grazing model (Appendix G).

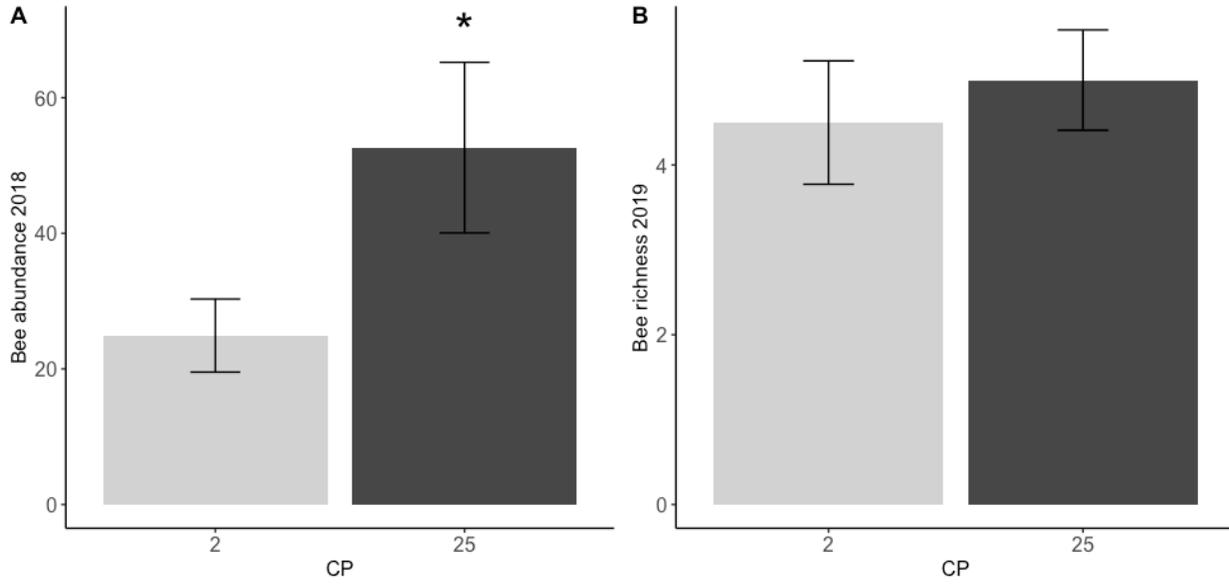
There was a stark contrast in parameters included in top model sets between years: for models estimating 2018 bee abundance, genus-richness, and diversity, ecoregion was rarely

**TABLE 2.** Estimates ( $\beta$ ) for parameters with detectable effects on each bee community response metric (abundance, richness, and diversity) in each year of the study (2018 and 2019). Effects were qualified as ‘detectable’ if 85% confidence intervals (CI) for parameter estimates did not overlap zero. Estimates and CI’s were standardized by model parameter partial standard deviations. For grassland ecoregions, “Mix”, “Short”, “Tall” refer to mixed-grass, short-grass, and tall-grass prairies (respectively), and “Sand” refers to sand prairie. Bolded estimates indicate predictors or levels of predictors with the greatest magnitude of effect, as determined by absolute-value distances of the upper or lower confidence interval (sign dependent) from 0.

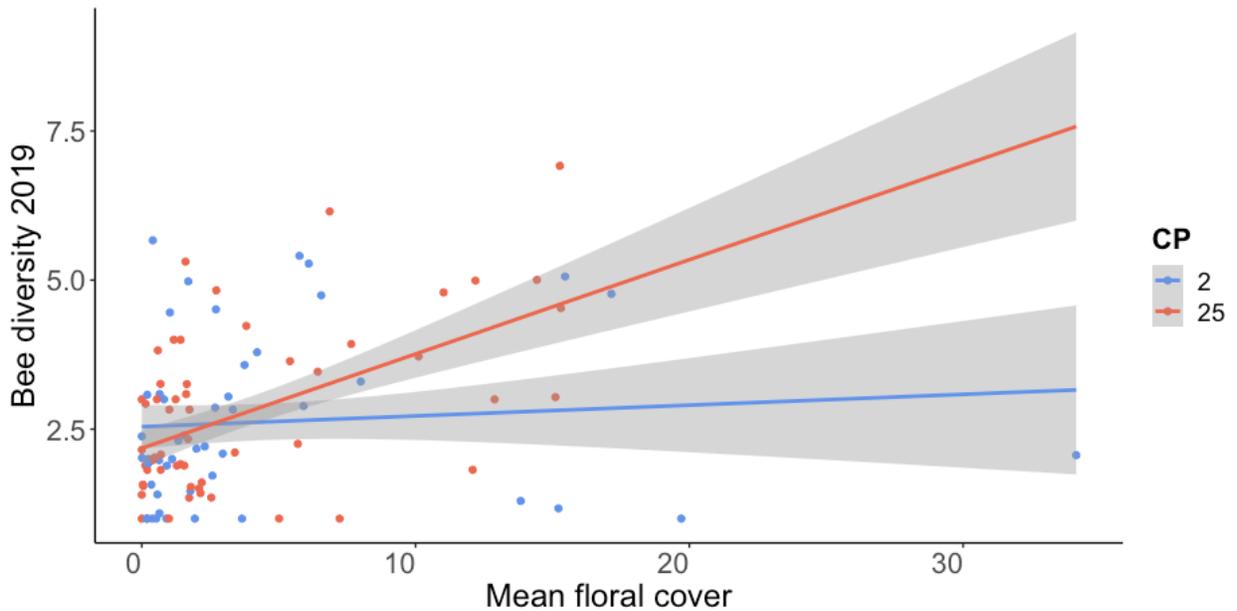
Year	Parameter	Bee abundance	Bee richness	Bee diversity	
		$\beta \pm 85\% \text{ CI}$	$\beta \pm 85\% \text{ CI}$	$\beta$	
2018	Floral cover	0.13 (0.09, 0.16)	0.06 (0.04, 0.08)	0.13 (0.02, 0.24)	
2019				0.11 (0.06, 0.16)	
2018	Floral richness			0.11 (0.023 0.20)	
2018	CP-25 v. CP-2	0.59 (0.31, 0.09)			
			0.31 (0.06, 0.56)		
2019	Ungrazed v. Grazed	-0.72 (-1.17, -0.21)		1.72 (0.69, 2.75)	
	Ecoregion	Sand v. Mix	-2.09 (-2.88, -1.30)		
		Tall v. Mix	-1.67 (-2.33, -1.01)		1.59 (0.85, 2.44)
		Short v. Sand	<b>2.62 (1.93, 3.12)</b>	0.57 (0.12, 1.02)	
		Tall v. Sand			0.93 (0.02, 1.84)
		Tall v. Short	-2.20 (-2.74 -1.66)	-0.42(-0.70, -0.15)	1.42 (0.73, 2.12)
		CP x Floral cover			0.06 (0.01, 0.11)
		CP-2			
		CP-25		0.14 (0.07, 0.21)	
	CP x Grazing		-0.40(-0.76, -0.05)		
		CP-2			
	Ecoregion * Grazing	Sand v. Mix		-1.41 (-2.08, -0.73)	-2.37 (-3.72, -1.02)
		Short v. Mix		-0.56 (-1.60, -0.06)	-1.36 (-2.54, -0.18)
		Tall v. Mix		-0.70 (-1.20, -0.20)	<b>-2.34 (-3.61, -1.06)</b>
Tall v. Sand			<b>0.84 (0.24, 1.45)</b>		
Short v. Sand			0.71 (0.10,1.31)		
Tall v. Short				-0.97 (-1.92, -0.03)	

included as a parameter in any preliminary candidate models with  $\Delta AICc < 4$  and subsequently did not occur in any model-averaged sets (Appendix H). Additionally, the effects of grazing on bee communities were negligible in 2018, as 85% confidence intervals for grazing parameter estimates overlapped zero (Appendix J). Floral cover was included in all 2018 model-averaged models and had a positive effect on abundance, richness, and diversity (Appendix I, Table 2); unlike 2019 however, neither floral resource metric was collinear with grazing. Conversely in 2019, floral cover and richness were collinear with both ecoregion *and* grazing; grazing and ecoregion parameters were ubiquitous across all models averaged, and bee communities had detectable responses to grazing and ecoregion.

Forb-enhanced restorations (CP-25) did not have the effects on bee communities when compared with grass plantings (CP-2) as initially predicted and CP-specific effects were only detectable for bee abundance in the first year (2018) and bee richness in the second year. This was the case despite that the parameter for restoration type (CP) occurred in every top model set across both years (Appendix I). Changes in bee abundance and richness were positively associated with CP-25 (Table 2), and more bees (Fig. 5 A, 2018) and more bee genera (Fig. 5 B, 2019) were collected on CP-25 fields. The effect of restoration type on bee richness was marginal however – 85% confidence intervals for mean richness overlapped (Fig. 5 B). Additionally, 2019 bee diversity response to floral cover differed between CP-2 and CP-25 plots (Table 2, Fig. 6). Although bee diversity was positively associated with mean floral cover on both forb-enhanced and grass plantings, the magnitude of the effect of floral cover on 2019 bee diversity was much greater on forb-enhanced (CP-25) fields than on CP-2 grass plantings (Table 2, Fig. 6).

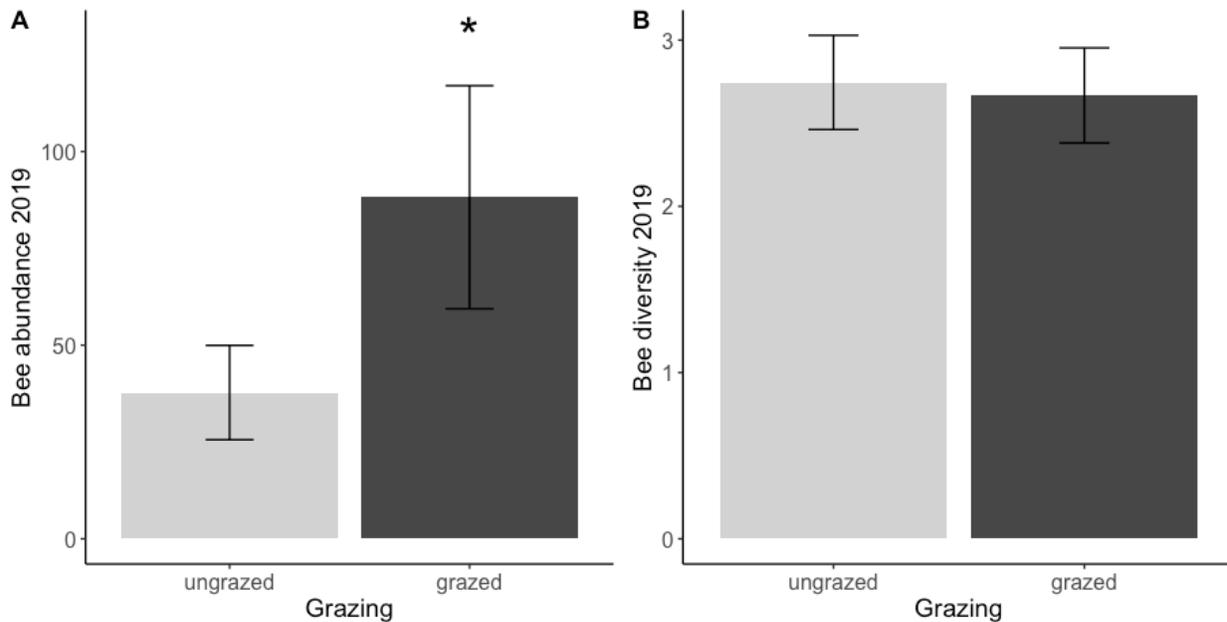


**FIG. 5** Mean bee abundance (2018, A) and bee richness (2019, B) for CP-2 (light gray) and CP-25 (dark gray) restorations. Error bars represent 85% confidence intervals and non-overlapping error bars are indicated by an asterisk (\*).

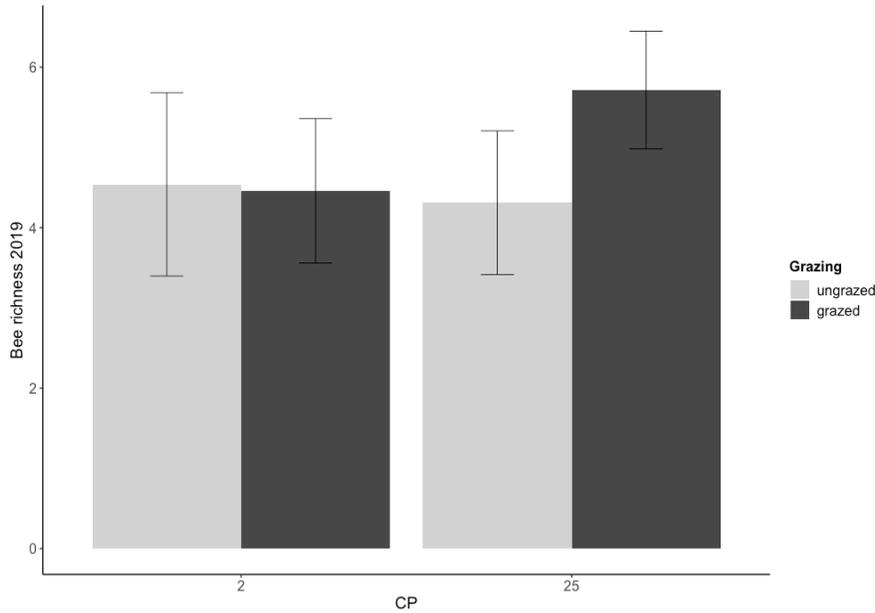


**FIG. 6** 2019 bee diversity (effective genera number) relative to mean quadrat percent floral cover and restoration type. CP-2 sites are shown in blue and CP-25 in red. Best fit lines are accompanied by 85% confidence intervals in gray.

Bee responses to grazing, independent of interactions, were detectable for bee abundance and bee diversity one year post-grazing (2019; Table 2, Fig. 7). Although grazing-associated effects were greatest in overall magnitude compared to all other parameters estimating bee diversity (Table 2), the difference between grazed and ungrazed fields was largest for bee abundance (Fig. 7 A), and grazed fields supported significantly more bees than ungrazed fields. Unlike bee abundance, bee diversity was somewhat negatively affected by grazing, though the difference between grazed and ungrazed fields was not large (Fig. 7 B). Bees did not respond uniformly to grazing between CP-25 and CP-2 plots in 2019, and again, this difference was detectable for bee richness (Table 2), which did not differ between grazed and ungrazed grass plantings (CP-2) but did increase in response to grazing on CP-25 fields relative to CP-2 (Fig. 8).



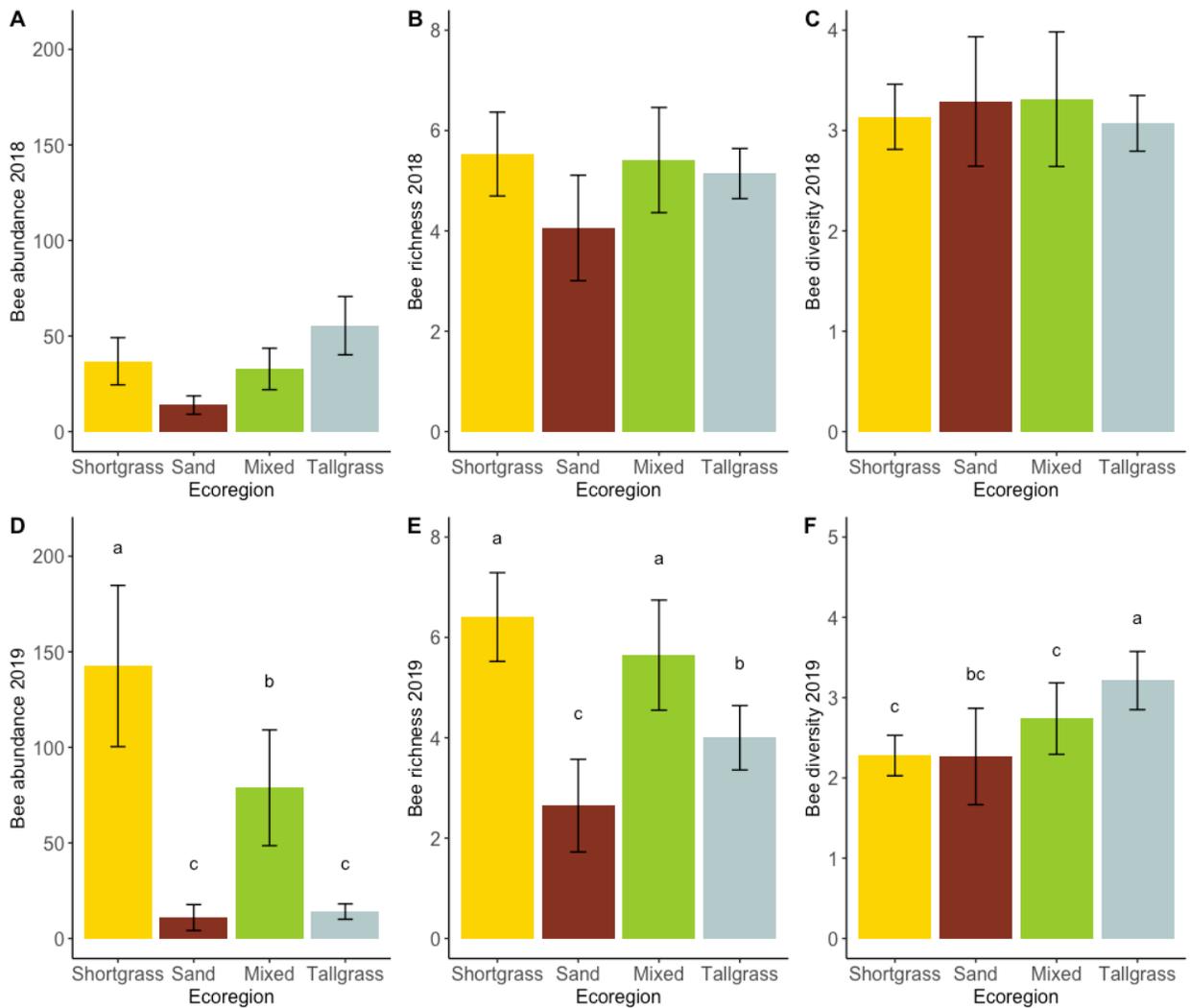
**FIG. 7** Bee abundance (A), bee diversity (B) for grazed (dark gray) and ungrazed (light gray) treatments. Bar heights indicate mean response, and error bars give 85% confidence intervals for these means. The asterisk above mean gray bee abundance indicates non-overlapping error bars.



**FIG. 8** Mean 2019 bee genus-richness for ungrazed (light gray) and grazed (dark gray) plots on grass-specific restorations (CP2, left) compared with forb-specific restorations (CP25, right) in 2019. Error bars indicate 85% confidence intervals for mean richness.

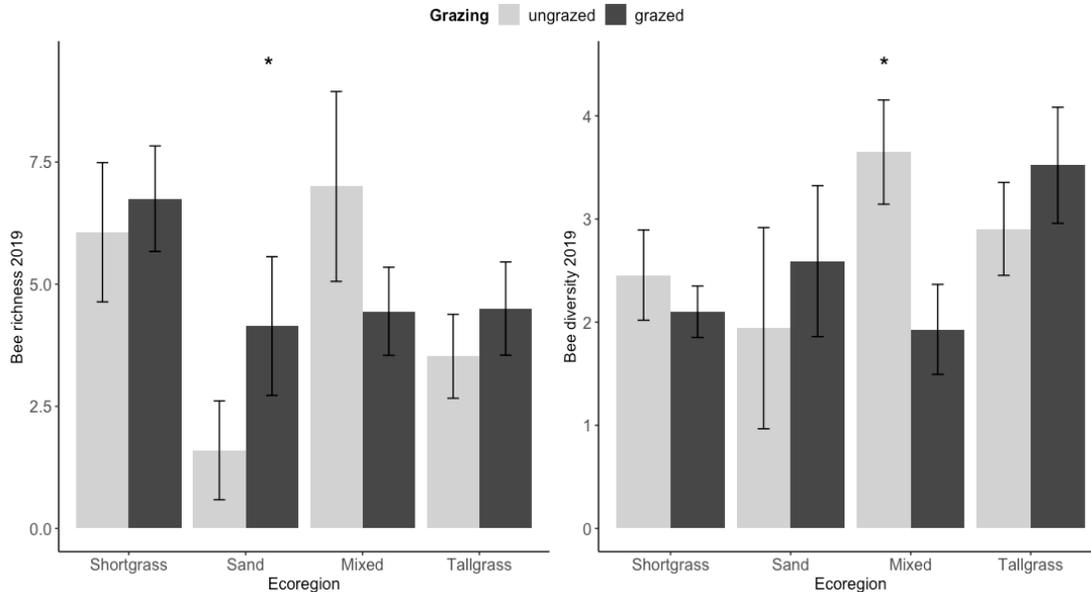
Given the broad longitudinal gradient of this study across the grassland landscape of Kansas, ecoregional differences allow us to understand variation in biotic and abiotic factors that may have widespread effects, including on bee communities. The effect of ecoregion (sand prairie and short-, mixed-, or tall-grass prairies) was detectable across all univariate bee community metrics in 2019 (Fig. 9), and ecoregional grazing effects were detectable for bee richness and bee diversity (Fig. 10, Table 2). Short-grass prairie fields supported more bee genera than any other ecoregion in 2019, and bee richness was lowest overall on sand prairie fields (Fig. 9 E). Alternatively, bees in the tall-grass ecoregion were more diverse than all other ecoregions (Fig. 9 B).

The combined effects of ecoregion and grazing were detectable for richness and diversity in 2019 (Table 2), which were, except for mixed-grass fields and short-grass prairie bee diversity, positively associated with, or not effected by, grazing compared with ungrazed fields (Fig. 10). Sand prairie bee richness in particular was notably higher on grazed plots (Fig. 10 A).



**FIG. 9** 2018 (top) and 2019 (bottom) bee abundance (A, D), richness (B, E) and diversity (C, F) for short-grass (yellow), sand (red), mixed-grass (green), and tall-grass (blue) prairie ecoregions. Bar heights indicate means, and 85% confidence intervals around these means are denoted with error bars. For 2019 plots, letters indicate differences between ecoregions with 85% confidence intervals not containing zero (Table 2). No letters are provided for 2018 plots because ecoregion was not included in model-averaged model sets.

In the mixed-grass prairie ecoregion however, bee richness was highest on ungrazed fields (Fig. 10 A), as was bee diversity (Fig. 10 B). Additionally, short-grass prairie bee diversity was somewhat higher on ungrazed fields than those that were grazed. All pairwise comparisons of ecoregional grazing effects on bee richness and diversity combined were detectable (Table 2).



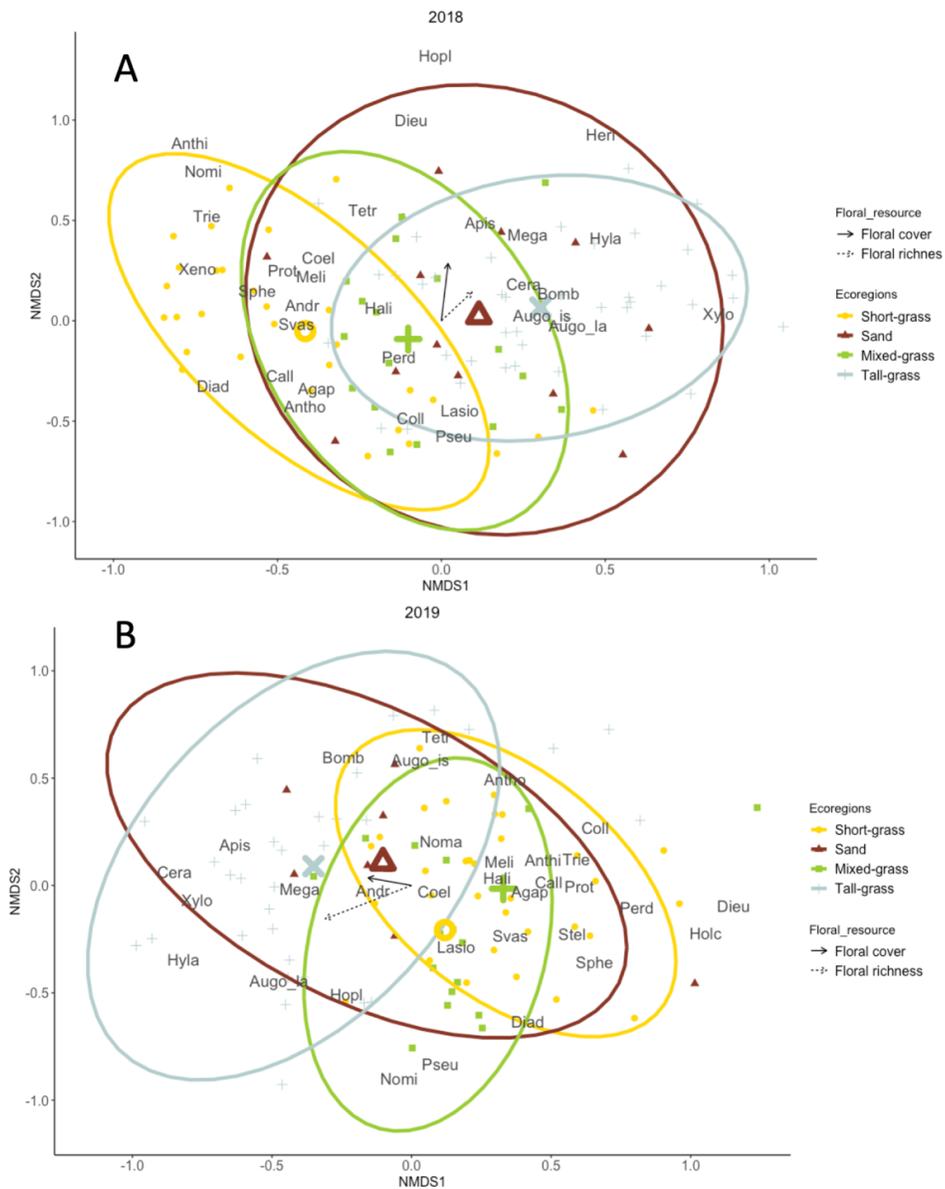
**FIG. 10** Differences in univariate bee measures of richness (A) and diversity (B) between grazed (light gray) and ungrazed (dark gray) fields in each ecoregion in this study. The height of the bars corresponds to means for respective bee metrics and treatment levels. Error bars indicate 85% confidence intervals around the mean. Asterisks (\*) indicate where 85% confidence intervals do not overlap within ecoregions between ungrazed and grazed groups.

### *Bee community comparisons*

In addition to multimodel inference, we conducted multivariate permutational analyses of variance (PERMANOVA) to determine if our primary treatment groups of ecoregion, restoration type (CP), and grazing significantly affected CRP bee community composition. Of these, only grassland ecoregion (short-grass, sand, mixed-grass, and tall-grass prairies) had a significant effect on bee community composition, which was the case in both 2018 and 2019 (PERMANOVA: 2018 pseudo- $F_{3,89}=4.81$ ,  $p=0.001$ ; 2019 pseudo- $F_{3,76}=8.66$ ,  $p=0.001$ ; Fig. 11). *Post-hoc* pairwise comparisons indicated that bee communities differed significantly between all ecoregions across years, save for mixed- and short-grass prairies ( $t_{45}=1.21$   $p=0.172$  for 2018;  $t_{46}=1.22$   $p=0.159$  for 2019). Additionally, in 2018, PERMANOVA revealed a marginally significant effect (95% confidence) of the interaction between restoration type (CP-2 and CP-25) and ecoregion on bee community composition (Appendix K and M, pseudo- $F_{3,89}=4.81$ ,  $p=0.001$ ).

There were significant differences in the bee communities found on CP-25 fields among all pairwise comparisons of ecoregions except mixed- vs. short-grass prairies ( $t_{22}=1.2644$ ,  $p=0.17$ ). CP-2 bee communities were less variable in composition among ecoregions, differing significantly only between tall-grass and short-grass prairies ( $t_{28}=1.9378$ ,  $p=0.001$ ) and between tall-grass and mixed-grass prairies ( $t_{21}=1.4021$ ,  $p=0.049$ ). Bee community composition was not significantly affected by grazing in either year of the study (Appendix K).

Multiple bee genera in 2018 and 2019 were significant indicators of each grassland ecoregion (Appendix L) and likely contributed to the differences between bee communities. Of these indicator genera, each ecoregional bee community (except for sand prairie bees) was significantly indicated by at least one genus across *both* years of the study: *Tripeolus* in short-grass prairies ( $t=0.274$ ,  $P=0.027$ , for 2018 and  $t=0.402$ ,  $P=0.011$  for 2019), *Pseudopanurgus* in mixed-grass prairies ( $t=0.340$ ,  $P=0.012$  for 2018,  $t=0.274$ ,  $P=0.013$  for 2019), and *Hylaeus* in tall-grass prairies ( $t=0.359$ ,  $P=0.023$  for 2018 and  $t=0.337$ ,  $P=0.01$  for 2019). Genera significantly indicative of sand prairie fields were *Apis* (honeybees) in 2018 ( $t=0.269$ ,  $P=0.038$ ) and *Xylocopa* (carpenter bees) in 2019 ( $t=0.399$ ,  $P=0.002$ ). Additionally, *Lasioglossum* was a significant indicator genus of both mixed- and short-grass CRP fields in 2018 ( $t=0.301$ ,  $P=0.034$ ) and 2019 ( $t=0.435$ ,  $P=0.009$ ).



**FIG. 11** NMDS ordinations using Bray-Curtis distances for 2018 (A) and 2019 (B) bee communities, grouped by short-grass (gold), mixed-grass (green), sand (red), and tall-grass (blue) prairie ecoregions. Floral resource metrics (floral species richness and floral cover) are plotted as vectors for visualization purposes, but significance levels are not provided due to high stress of the ordinations in both years. Abbreviations for genera are as follows: **Ag**a=Agapostemon, **Andr**=Andrena, **Anth**=Anthophora, **Augo\_la**=Augochlorella, **Augo\_is**=Augochloropsis, **Bomb**=Bombus, **Call**=Calliopsis, **Cera**=Ceratina, **Coel**=Coelioxys, **Coll**=Colletes, **Diad**=Diadasia, **Dieu**=Dieunomia, **Hali**=Halictus, **Heri**=Heriades, **Hopl**=Hoplitis, **Hyla**=Hylaeus, **Lasio**=Lasioglossum, **Mega**=Megachile, **Meli**=Melissodes, **Nomi**=Nomia, **Perd**=Perdita, **Prot**=Protandrena, **Pseu**=Pseudopanurgus, **Sphe**=Sphecodes, **Svas**=Svastra, **Tetr**=Tetraloniella, **Trie**=Triepeolus, **Xeno**=Xenoglossa, **Xylo**=Xylocopa.

## DISCUSSION

This study is the first of its kind to systematically document bee communities and assess restoration management practices across a broad landscape (650 km and a longitudinal precipitation gradient of 63.5 cm) that encompasses four grassland ecoregions of the Great Plains. Our findings provide critical insight into bee community responses associated with common restoration management practices implemented on retired agricultural land. Additionally, bees collected in this study contribute to baseline estimations of native bee diversity throughout a substantial portion of the Great Plains. This baseline is essential for monitoring changes in bee community diversity and abundance and subsequently informing effective land management and conservation decisions.

We assessed the effects of restoration type (forb-enhanced vs. grass plantings) and a grazing disturbance regime implemented on restored grasslands and found a majority of individuals belonged to only a few, dominant genera, which reflects similar patterns observed in studies of species-level bee communities (Williams et al. 2010, Potts et al. 2003). Of the 40 genera collected in this study, 20% were rare, i.e. represented by only one or two individuals (Appendix E), and 80% of all individuals belonged to three dominant genera: *Lasioglossum*, *Augochlorella*, and *Agapostemon*. The relatively high percentage of rare bee genera found in our study (20%) is similar to other studies conducted on restored CRP grasslands that found 11 – 45% of bee genera to be represented by one or two individuals (Arathi et al. 2019 and Haley et al. 2018, respectively). Given the history of disturbance typical of CRP lands, the presence of ‘rare’ genera on these restorations may be explained by occurrences of transient, non-resident bees temporarily utilizing floral resources between more suitable nesting or foraging patches.

### *Summary of bee responses*

Compared to the first year of this study (2018), we collected nearly 2,000 more bees following the discontinuation of cattle grazing on CRP fields in 2019 (Appendix E). Additionally, key factors driving bee communities, as identified via information-theoretic approaches, were markedly different between the two years of the study. Restoration type (CP) had some positive effects of bee community measures, but these effects were not universally detectable across years or bee response measures (Table 2, Fig. 5). Additionally, metrics of floral resource availability (floral cover and floral species richness) did not differ significantly between forb-enhanced restorations and grass plantings (Figs. 3 G-J, Appendix F). Responses of both bee and floral communities to grazing disturbances were mostly positive, though not detectable until one year post-grazing (Table 2, Figs. 3 C and F, Fig. 7 A, Figs. 10 A–B). Floral cover was positively related to bee abundance and richness in the first year, and effective genera number (herein ‘diversity’) in both years (Table 2). Floral species richness however, only had detectable effects on post-grazing (2019) bee diversity (Table 2). Trends in floral resources were paralleled closely by trends in native bee responses to grazing at the ecoregional level (Fig. 10, Fig. 4), and bee community composition differed significantly among ecoregions in both years (Fig. 11). Ultimately, ecoregional and grazing-induced differences in floral resource availability were the primary drivers identified in our analyses, with detectable direct and indirect effects on all bee community measures.

### *Treatment effects: restoration type*

Despite differences between CP-25 and CP-2 average age, management history, and initial plantings, we found that restoration type did not have consistently detectable effects on the

abundance, richness, diversity, and composition of native bee communities across years, and did not significantly affect bee diversity in either year of the study. Bee abundance was greater on forb-enhanced (CP-25) fields in the first year of the study, but not the second (Fig. 5 A). Bee richness was somewhat positively associated with CP-25 fields in the second year of the study, but there was no detectable influence of restoration type in the first year (2018; Fig. 5 B). Overall, forb-enhanced restorations did not have the strong, ubiquitously positive effects on native bee communities that we expected. However, our prediction that bee responses to CP would be mediated through differences in floral resource availability was at least indirectly supported in that restoration type was not correlated with floral cover or richness, and thus did not have comprehensively detectable effects on bee communities.

Although bee abundance, richness, and diversity were not largely responsive to restoration type, these univariate metrics may be insufficient for capturing functional differences in bee communities. Multivariate analyses revealed differences in ecoregional bee community composition between CP-2 and CP-25 restorations in the first year of the study (Appendix K, Appendix M). In general, bee communities on CP-2 grass plantings were more homogenous in composition than CP-25 fields (Appendix M). That no compositional differences were detected in 2019 (Appendix K) could be due to the necessary elimination of singleton genera prior to analyses. Although there was substantial overlap in the occurrence of 75% of all genera between restoration types, 10 bee genera were unique to (i.e. found only on) forb-enhanced restorations (CP-25) while only two genera were unique to grass plantings. These genera were all collected in low abundances and were primarily comprised of a) specialist nesters, i.e. aboveground cavity-nesting *Ashmeadiella* and *Heriades* on CP-25, *Dianthidium* on CP-2, b) cleptoparasites, i.e. *Epeolus*, *Xeromelecta*, and *Stelis* on CP-25 and *Holcopasites* on CP-2, and c) floral specialists:

i.e. Asteraceae oligolege *Dieunomia* and Cucurbita oligolege *Xenoglossa* (Michener 2007) on CP-25. In general, aboveground nesters, cleptoparasites, and floral specialists (oligoleges) are most affected by anthropogenic disturbance and habitat fragmentation compared to bees with generalized life-history traits (Williams et al. 2010, Sheffield et al. 2013, Winfree et al. 2011). Thus CP-25 fields may confer benefits to specialized bee taxa that CP-2 fields do not.

Although univariate floral responses did not differ by restoration type, observed bee community compositional differences between CP-2 and CP-25 fields (Appendix F, Fig. 3 G-J) may be affected by floral community variation not captured in metrics of percent cover and species richness. Bee communities are structured by the combined variation in floral species traits including, but not limited to: native status or endemism (Stout and Morales 2009, Bjercknes et al. 2007, Tepedino et al. 2008, Denning et al. 2018), bloom time and duration (Dählstrom et al. 2008), annual vs. perennial growth form (Potts et al. 2003), pollen-quantity-to-nectar-quality ratio (Potts et al. 2003), and pollinator associations. Although we did not specifically address differences in flowering forb community composition or floral species traits in these analyses, we did observe distinct differences in the dominant forb species on CP-2 and CP-25 fields. In both years, the introduced Fabaceae species *Melilotus officinalis* and *M. alba* were dominant in the floral communities of grass-planted (CP-2) fields. *Melilotus* percent cover in 2018 was more than three times that of the next most abundant species, and the difference was more than seven-fold in 2019. On CP-25 fields, *Melilotus* species also had the highest percent cover across plots but were, at most, only twice as abundant as multiple other species with similarly high cover, including the native prairie forb species *Dalea purpurea* and *Ratibida pinnata*. Although non-native pollen sources have been identified as potentially valuable resources for native pollinators (Bjercknes et al. 2007, Tepedino et al. 2008), there is evidence that bees prefer native flowers over

exotics (Williams et al. 2011) and that this preference is strongest for specialist bees (Memmott and Waser 2002, Frankie et al. 2005). This may explain both a) the positive association between bee abundance and richness and CP-25 fields in 2018 and 2019, respectively and b) the difference in magnitude of the strength of bee diversity-floral cover relationships on CP-2 and CP-25 fields, wherein bee diversity on forb-enhanced (CP-25) fields was more positively related to floral cover than was bee diversity on grass-plantings (CP-2, Fig. 6). Given that the introduced *Melilotus* species comprised a majority of overall floral cover on CP-2 fields, increased *Melilotus* cover on grass plantings may not necessarily attract more bee species; on CP-25 however, increased cover of native species may be more likely to attract a greater diversity of bees. Therefore, while the overall floral cover and species richness of CP-2 and CP-25 fields were comparable, the quality and evenness of floral resources possibly plays an important role in structuring bee communities.

#### *Treatment effects: Grazing*

Overall, the effects of grazing on both bee communities and floral resource availability were generally positive, though only detected in the second year of the study (2019; Table 2), when cattle were no longer actively grazing. Bee abundance increased on grazed plots (Fig. 7 A), but there was a weak negative effect on bee diversity (Fig. 7 B). Similarly, floral resource measures were positively correlated with grazing in 2019 but were not significantly different between grazed and ungrazed plots the prior year. Thus our prediction that bee communities would respond positively to grazing was partially true in that 2019 bee abundance was highest on grazed fields; however, expected increases in bee diversity did not follow observed increases in floral cover and richness (Figs. 3 A and B) and the effects of grazing on bee richness overall

were not detectable (Appendix I). Additionally, we expected there would be ecoregional differences in bee responses to grazing and found that grazing-effects were indeed pronounced at the ecoregional level for 2019 bee richness and diversity (Fig. 10). In every ecoregion except mixed-grass prairie, more bee genera occurred on grazed plots compared with ungrazed plots (Fig. 10 A). Mixed-grass bee richness and diversity, however, were negatively associated with grazing, as was bee diversity in the short-grass prairie ecoregion, to a lesser extent (Fig. 10 B).

Although bee community composition did not differ significantly between ungrazed and grazed fields (Appendix K), differences in indicative bee genera followed those observed by other studies, indicating potentially predictable responses of certain bee taxa to grazing. The semi-social, ground-nesting genus *Lasioglossum* was a significant indicator of grazed fields (Appendix L) and was approximately three times as abundant compared with ungrazed fields. This trend is not unique to our study. For example, *Lasioglossum* abundance was positively associated with cattle grazing on a Mediterranean nature reserve (Vulliamy et al. 2006). In another study, *Lasioglossum* abundance was not affected by grazing intensity in Pacific Northwest bunchgrass prairies (Kimoto et al. 2012). This resiliency of *Lasioglossum* to grazing disturbances might be explained by grazing-induced changes in soil properties. Ground-nesting sweat bees like *Lasioglossum* typically prefer nesting sites with more bare ground and compacted soils (Potts and Willmer 1997, 1998), potentially resulting in *Lasioglossum* affinity to grazed areas due to favorable effects of grazing on nesting-site quality. In these same studies and others, many bumblebee species (genus *Bombus*) had opposite responses to grazing (Kimoto et al. 2012, Hatfield and Lebuhn 2007, Sjödin 2007), and decreased in abundance and diversity due to the removal of preferred floral resources (Xie et al. 2008) and the destruction of nesting habitat (Kimoto et al. 2012). Because body size is directly correlated with foraging range

(Greenleaf et al. 2007) and is typically associated with dispersal ability (Carrié et al. 2017), *Bombus* are thus able to leave a given patch for the opportunity to find diverse floral resources elsewhere (Jha and Kremen 2013). While *Bombus* specifically was not significantly indicative of ungrazed plots in this study, bees in the genus *Svastra* were. Similar to *Bombus*, *Svastra* are large-bodied bees that nest below ground (Cane 1995). To the best of our knowledge, the response of *Svastra* to grazing has not been explicitly examined; however, it is likely that, like *Bombus*, *Svastra* are similarly affected by a) the removal of floral hosts and b) decreased quality of nesting habitat. *Svastra* are oligoleges of Asteraceae (Cane 2017) and one common prairie species, *Svastra obliqua*, is known for its associations with *Helianthus* (sunflowers). Grazing can have negative impacts on *Helianthus* densities and cause premature senescence (Reece et al. 2004), in addition to having negative effects on the development of other Asteraceae species (Hickman and Hartnett 2002). In 2019, while we did not observe any flowering *Helianthus* individuals on grazed fields, *H. annuus* and *H. petiolaris* were both present and flowering on ungrazed fields. Given this, and *Svastra*'s large body size, it is possible that bees in this genus dispersed from grazed plots to habitats with greater availability of preferred floral resources. Finally, because *Svastra* nest below ground (Rozen 1983) but do not typically display a preference for compacted soils, cattle-induced changes to soil properties might decrease the suitability of nesting habitat and additionally motivate dispersal.

That bee community responses to grazing were only detectable in 2019 may be a result of latency in bee community responses to the effects of grazing-induced floral community changes during the first (2017) and second (2018) years of grazing. In this context, latent effects, also known as 'lag-effects' or 'pulsed' relationships, would suggest that bees in a given year are more highly correlated with the resources of the previous year than the year in which they were

observed (Tepedino and Stanton 1981). The extent to which lag effects drive bee communities is unclear. Blaauw and Isaacs (2014) observed increased wild bee abundance in agroecosystems 2-3 years *after* newly planted diverse and abundant wildflower communities successfully established. Another study found latency to be taxa-specific: *Anthophora* bee abundance increased in the year following a year of abundant floral resources, while *Bombus* abundance did not (Crone 2013). Although in our study grazing was not correlated with floral measures in 2018, grazers have been shown to affect resource availability in ways not measurable by floral richness or diversity. Grazing can differentially affect forb species (Hartnett et al. 1996, Hickman and Hartnett 2002), thus altering the availability of certain floral resources, which may also vary in pollen and nectar quality (Kearns and Inouye et al. 1997). Nectar and pollen quantity, quality, and diversity play an essential role in determining bee developmental success (Filiapak et al. 2017). In Iowa and Missouri grasslands, one study found cattle stocking rates to be negatively associated with bee nutritional indicators such as lipid content and body size, which relate directly to nutritional stress during development (Smith et al. 2016). On the other hand, there is evidence that the grazing of grasses preferred by cattle can result in competitive release and lead to increased forb reproduction (Damhoureyeh and Hartnett 1997). If bee community responses to the effects of low-intensity grazing are indeed latent, it is possible that differences in bee abundance, richness, and diversity on grazed vs. ungrazed plots in 2019 reflect grazing-induced changes to floral resources in the previous year that are not measurable by metrics of floral cover and richness associated with the current year.

Lag effects of previous-year floral communities on 2019 bee communities may also be driven by extreme weather events across the study area in 2018. For example, below-average rainfall in 2018 throughout the tall-grass prairie ecoregion led to severe drought during the

period of time in which we sampled CRP fields. Tall-grass prairie plants are shown to have decreased growth during instances of drought and in the presence of grazers (Booth 1941, Weaver 1954, respectively). In the year following the drought, bee abundance on tall-grass fields was reduced to 25% that of 2018 levels (Figs. 9 A and D). However, floral cover increased simultaneously. Therefore, changes in flowering forb growth due to limited water availability likely had negative effects on bee fecundity in 2018, resulting in lower brood sizes and thus decreased abundance of 2019 bees. In general, the effects of a disturbance like grazing on grasslands are typically secondary to the effects of increased or decreased water availability (Milchunas and Lauenroth 1993, Biondini et al. 1998). Thus, discerning between the effects of grazing and climatic drivers proves problematic when conditions are variable. However, these conditions likely arise more often than not when land managers apply management regimes to restored grasslands, making weather and grazing effects mutually inclusive when it comes to devising best-practice management regimes in the real world.

### *Management implications*

Agricultural intensification is expected to increase with global demands for food (Godfray et al. 2010), putting grassland agroecosystems at further risk for biodiversity declines, habitat fragmentation, and habitat loss. In light of these threats, grassland restoration efforts like the Conservation Reserve Program have expanded over the past 20 years to include the establishment of wildlife and pollinator habitat as primary conservation goals. Ideally, successfully restored lands support diverse and abundant forb communities and, subsequently, robust pollinator communities. Prior to this study, however, very little research had been conducted to a) verify that restorations identified as beneficial to pollinators actually provide

sufficient pollinator habitat within agroecosystems and b) confirm that associated management practices align with CRP conservation objectives and contribute to the stability of pollination and other ecosystem services.

Altogether we found slightly positive, albeit weak, effects of forb-enhanced restorations (CP-25) on bee abundance and genus-richness (Fig. 5). CP-25 fields had high richness of unique bee genera and greater relative abundance of native vs. introduced dominant forbs than did grass plantings (CP-2). These results indicate that forb-enhanced restorations targeting the establishment of pollinator habitat may confer benefits to bee communities beyond those readily identified by univariate measures of bee and floral communities.

Despite the inclusion of two- to five-times the number of forb species in initial seed mixes, CP-25 fields did not have greater floral cover or floral richness compared with CP-2 (Figs. 3 G-J). The importance of floral resource abundance (i.e. percent cover) for all measures of native bee communities was well-supported by our results, and the positive associations of floral richness and diversity with bee abundance, richness, and diversity have been documented across numerous studies (e.g. O'Toole 1993, Potts et al. 2003, Hopwood 2008, Garrido et al. 2019). Therefore, increasing forb establishment on these forb-enhanced restorations is an important step in restoring pollinator habitat capable of supporting diverse and abundant bee and pollinator communities. On CRP land, failure of forb establishment has been attributed to competition with dominant grass species (McCain et al. 2011). Increasing establishment success might be achieved by increasing forb density in initial seeding mixes, or, given high costs of forb seeds, reducing the relative densities of grass seeds (Dickson and Busby 2009).

Grazing is an essential ecological driver of the Great Plains (Fuhlendorf and Engle 2001) and thus an essential component of restored grassland management. Our results suggest that low-

to-medium intensity cattle grazing on grassland restorations may benefit bee abundance and diversity (Fig. 7). The effects of grazing were most evident at the ecoregional scale, revealing largely positive responses except for decreased bee richness and diversity on grazed mixed-grass restorations, as well as marginal decreases in bee richness on grazed short-grass fields (Fig. 10). That these effects were only observed in bee communities the year *after* discontinuation of grazing (2019) suggests generationally latent bee community responses to grazing-induced floral community changes.

Determining optimal grazing regimes for CRP land requires a multi-faceted approach to understanding how grazing covaries with other habitat and environmental factors. We observed clear ecoregional differences in the direction and extent of bee responses to grazing, suggesting that making decisions regarding grazing of restored grasslands requires a localized approach. Sand prairie bees exhibited drastic increases in all response measures on grazed fields, suggesting that grazing may be a beneficial disturbance in this region. Alternatively, mixed-grass bee diversity decreased significantly on grazed fields compared with ungrazed fields. In both of these extreme instances, the mechanisms driving these differences are unclear. Historically, sand prairies were thought to be sensitive to over-grazing (Ramaley 1939), and in some conservation areas, grazing has been eliminated altogether (Platt 1973). The opposite appears true for CRP land in the sand prairie ecoregion of Kansas – with the caveat that grazing in this study was short-term and of low-intensity and that sand prairie CRP fields may not be sufficient surrogates of native prairies. The stocking rate implemented by participating landowners via NRCS recommendations was based on a target goal of 50% plant biomass reduction. In one study, this target was also identified as the optimal stocking rate to be employed on mixed-grass prairie fields, specifically (Biondini et al. 1998). In reality, overall biomass reduction reached only up to

33% or less in our study plots (D.Fraser Watson, *personal communication* on Oct. 25, 2019).

Although a number of studies have identified negative associations of grazing intensity with bee communities (e.g. Hatfield and LeBuhn 2007, Sjödin 2007, Xie et al. 2008), it seems unlikely that this is the case in our study. Accordingly, further research is necessary to assess the benefits and drawbacks of grazing on restored grasslands within the many ecoregions of the Great Plains.

Native bees are exceptionally diverse in their life-history traits and choosing ‘best’ management practices for the conservation of pollinator habitat and bee communities requires substantially more research and a least-harm approach. We know relatively little about: a) species-specific effects of grazing on flowering forb species and subsequent effects on foraging and dispersal of native bees; b) specific nesting biology and nesting habitat preferences of bees and subsequent benefits/costs conferred by management and disturbance regimes; and c) interactive effects of grazing and climatic variables on bee communities. Understanding the mechanisms driving native bee responses to land management is a critical component of effective conservation strategies. Overall, it is clear that bee community declines cannot be monitored or even detected without continued support for, and implementation of, systematic sampling efforts (Biesmeijer et al. 2006, Bartomeus and Winfree. 2013, Burkle et al. 2013, Marlin and LaBerge 2001), such as in this study. Given the broad longitudinal gradient and multiple ecologically distinct regions encompassed by this research, our findings provide valuable insight into how bee communities respond to land management within the Great Plains at both a landscape and ecoregional scale.

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## APPENDICES



APPENDIX B

CRP FIELD REPLICATES BY ECOREGION AND COUNTY

<b>Ecoregion</b>	<b>County</b>	<b>Replicates</b>
<b>Shortgrass</b>	Gove	6
	Lane	4
	Logan	6
	Scott	1
	Sheridan	3
	Thomas	6
	Wallace	6
	<b>Total</b>	<b>32</b>
<b>Mixed</b>	Barton	3
	Edwards	1
	Hodgeman	1
	Ness	4
	Pawnee	2
	Rush	4
	Trego	2
	<b>Total</b>	<b>17</b>
<b>Sand</b>	Pawnee	1
	Reno	6
	Rice	4
	Stafford	6
	<b>Total</b>	<b>17</b>
<b>Tallgrass</b>	Allen	3
	Anderson	1
	Bourbon	5
	Coffey	7
	Dickinson	1
	Ellsworth	1
	Geary	1
	Greenwood	2
	Linn	4
	Lyon	4
	Marion	1
	Morris	1
	Osage	2
	Rice	5
Wabaunsee	2	
Woodson	2	
	<b>Total</b>	<b>42</b>

APPENDIX C

CRP ECOREGION COMPARISONS: NRCS VS. EPA

CRP fields and ecoregion designations with NRCS rare and declining habitat types and EPA IV ecoregions. NRCS designations used for 11 fields (bold text, gray rows)

<b>Plot Code</b>	<b>County</b>	<b>EPA Ecoregion IV</b>	<b>EPA prairie designation</b>	<b>NRCS rare and declining habitat</b>	<b>Final designation</b>
C31	Barton	27b Rolling Plains and Breaks	Mixed	Mixed/Sand	Mixed
C32	Barton	27b Rolling Plains and Breaks	Mixed	Mixed/Sand	Mixed
C33	Barton	27b Rolling Plains and Breaks	Mixed	Mixed/Sand	Mixed
C24	Edwards	27b Rolling Plains and Breaks	Mixed	Mixed/Sand/Sandsage	Mixed
C27	Hodgeman	27b Rolling Plains and Breaks	Mixed	Mixed	Mixed
C22	Ness	27b Rolling Plains and Breaks	Mixed	Mixed	Mixed
C23	Ness	27b Rolling Plains and Breaks	Mixed	Mixed	Mixed
W03	Ness	27b Rolling Plains and Breaks	Mixed	Mixed	Mixed
W22	Ness	27b Rolling Plains and Breaks	Mixed	Mixed	Mixed
C25	Pawnee	27b Rolling Plains and Breaks	Mixed	Mixed/Sand	Mixed
C26	Pawnee	27b Rolling Plains and Breaks	Mixed	Mixed/Sand	Mixed
C12	Rush	27b Rolling Plains and Breaks	Mixed	Mixed	Mixed

APPENDIX C (continued)

<b>Plot Code</b>	<b>County</b>	<b>EPA Ecoregion IV</b>	<b>EPA prairie designation</b>	<b>NRCS rare and declining habitat</b>	<b>Final designation</b>
C13	Rush	27b Rolling Plains and Breaks	Mixed	Mixed	Mixed
C14	Rush	27b Rolling Plains and Breaks	Mixed	Mixed	Mixed
C21	Rush	27b Rolling Plains and Breaks	Mixed	Mixed	Mixed
W04	Trego	27b Rolling Plains and Breaks	Mixed	Mixed	Mixed
W06	Trego	27b Rolling Plains and Breaks	Mixed	Mixed	Mixed
C09	Pawnee	27c Great Bend Sand Prairie	Sand	Mixed/Sand	Sand
C04	Reno	27c Great Bend Sand Prairie	Sand	Tallgrass/Sand	Sand
C05	Reno	27c Great Bend Sand Prairie	Sand	Tallgrass/Sand	Sand
C06	Reno	27c Great Bend Sand Prairie	Sand	Tallgrass/Sand	Sand
C08	Reno	27c Great Bend Sand Prairie	Sand	Tallgrass/Sand	Sand
C34	Reno	27c Great Bend Sand Prairie	Sand	Tallgrass/Sand	Sand
C35	Reno	27c Great Bend Sand Prairie	Sand	Tallgrass/Sand	Sand
C07	Rice	27c Great Bend Sand Prairie	Sand	Tallgrass/Sand	Sand
C18	Rice	27c Great Bend Sand Prairie	Sand	Tallgrass/Sand	Sand
C19	Rice	27c Great Bend Sand Prairie	Sand	Tallgrass/Sand	Sand

APPENDIX C (continued)

<b>Plot Code</b>	<b>County</b>	<b>EPA Ecoregion IV</b>	<b>EPA prairie designation</b>	<b>NRCS rare and declining habitat</b>	<b>Final designation</b>
C20	Rice	27c Great Bend Sand Prairie	Sand	Tallgrass/Sand	Sand
C10	Stafford	27c Great Bend Sand Prairie	Sand	Mixed/Sand	Sand
C11	Stafford	27c Great Bend Sand Prairie	Sand	Mixed/Sand	Sand
C28	Stafford	27c Great Bend Sand Prairie	Sand	Mixed/Sand	Sand
C29	Stafford	27c Great Bend Sand Prairie	Sand	Mixed/Sand	Sand
C30	Stafford	27c Great Bend Sand Prairie	Sand	Mixed/Sand	Sand
C36	Stafford	27c Great Bend Sand Prairie	Sand	Mixed/Sand	Sand
<b>W05</b>	<b>Gove</b>	<b>27b Rolling Plains and Breaks</b>	<b>Mixed</b>	<b>Shortgrass</b>	<b>Shortgrass</b>
<b>W13</b>	<b>Gove</b>	<b>27b Rolling Plains and Breaks</b>	<b>Mixed</b>	<b>Shortgrass</b>	<b>Shortgrass</b>
<b>W14</b>	<b>Gove</b>	<b>27b Rolling Plains and Breaks</b>	<b>Mixed</b>	<b>Shortgrass</b>	<b>Shortgrass</b>
<b>W15</b>	<b>Gove</b>	<b>27b Rolling Plains and Breaks</b>	<b>Mixed</b>	<b>Shortgrass</b>	<b>Shortgrass</b>
<b>W17</b>	<b>Gove</b>	<b>27b Rolling Plains and Breaks</b>	<b>Mixed</b>	<b>Shortgrass</b>	<b>Shortgrass</b>
<b>W18</b>	<b>Gove</b>	<b>27b Rolling Plains and Breaks</b>	<b>Mixed</b>	<b>Shortgrass</b>	<b>Shortgrass</b>
<b>W01</b>	<b>Lane</b>	<b>27b Rolling Plains and Breaks</b>	<b>Mixed</b>	<b>Shortgrass/Sandsage</b>	<b>Shortgrass</b>
<b>W02</b>	<b>Lane</b>	<b>27b Rolling Plains and Breaks</b>	<b>Mixed</b>	<b>Shortgrass/Sandsage</b>	<b>Shortgrass</b>

APPENDIX C (continued)

<b>Plot Code</b>	<b>County</b>	<b>EPA Ecoregion IV</b>	<b>EPA prairie designation</b>	<b>NRCS rare and declining habitat</b>	<b>Final designation</b>
W23	Lane	25d Flat to Rolling Plains	Shortgrass/mixed	Shortgrass/Sandsage	Shortgrass
W24	Lane	25d Flat to Rolling Plains	Shortgrass/mixed	Shortgrass/Sandsage	Shortgrass
W25	Logan	25c Moderate Relief Plains	Shortgrass/mixed	Shortgrass	Shortgrass
W26	Logan	25c Moderate Relief Plains	Shortgrass/mixed	Shortgrass	Shortgrass
W27	Logan	27b Rolling Plains and Breaks	Shortgrass/mixed	Shortgrass	Shortgrass
W28	Logan	27b Rolling Plains and Breaks	Shortgrass/mixed	Shortgrass	Shortgrass
W30	Logan	25d Flat to Rolling Plains	Shortgrass/mixed	Shortgrass	Shortgrass
W31	Logan	25d Flat to Rolling Plains	Shortgrass/mixed	Shortgrass	Shortgrass
W16	Scott	25d Flat to Rolling Plains	Shortgrass/mixed	Shortgrass/Sandsage	Shortgrass
<b>W07</b>	<b>Sheridan</b>	<b>27b Rolling Plains and Breaks</b>	<b>Mixed</b>	<b>Shortgrass</b>	<b>Shortgrass</b>
<b>W08</b>	<b>Sheridan</b>	<b>27b Rolling Plains and Breaks</b>	<b>Mixed</b>	<b>Shortgrass</b>	<b>Shortgrass</b>
<b>W09</b>	<b>Sheridan</b>	<b>27b Rolling Plains and Breaks</b>	<b>Mixed</b>	<b>Shortgrass</b>	<b>Shortgrass</b>
W10	Thomas	25d Flat to Rolling Plains	Shortgrass/mixed	Shortgrass	Shortgrass
W11	Thomas	25d Flat to Rolling Plains	Shortgrass/mixed	Shortgrass	Shortgrass
W12	Thomas	25d Flat to Rolling Plains	Shortgrass/mixed	Shortgrass	Shortgrass

APPENDIX C (continued)

<b>Plot Code</b>	<b>County</b>	<b>EPA Ecoregion IV</b>	<b>EPA prairie designation</b>	<b>NRCS rare and declining habitat</b>	<b>Final designation</b>
W29	Thomas	25d Flat to Rolling Plains	Shortgrass/mixed	Shortgrass	Shortgrass
W35	Thomas	25d Flat to Rolling Plains	Shortgrass/mixed	Shortgrass	Shortgrass
W36	Thomas	25d Flat to Rolling Plains	Shortgrass/mixed	Shortgrass	Shortgrass
W19	Wallace	25d Flat to Rolling Plains	Shortgrass/mixed	Shortgrass	Shortgrass
W20	Wallace	25c Moderate Relief Plains	Shortgrass/mixed	Shortgrass	Shortgrass
W21	Wallace	25c Moderate Relief Plains	Shortgrass/mixed	Shortgrass	Shortgrass
W32	Wallace	25c Moderate Relief Plains	Shortgrass/mixed	Shortgrass	Shortgrass
W33	Wallace	25c Moderate Relief Plains	Shortgrass/mixed	Shortgrass	Shortgrass
W34	Wallace	25d Flat to Rolling Plains	Shortgrass/mixed	Shortgrass	Shortgrass
E28	Allen	40b Osage Cuestas	Tallgrass/oak hickory woodland	Tallgrass	Tallgrass
E30	Allen	40b Osage Cuestas	Tallgrass/oak hickory woodland	Tallgrass	Tallgrass
E31	Allen	40b Osage Cuestas	Tallgrass/oak hickory woodland	Tallgrass	Tallgrass
E29	Anderson	40b Osage Cuestas	Tallgrass/oak hickory woodland	Tallgrass	Tallgrass
E22	Bourbon	40c Wooded Osage Plains	Tallgrass/oak hickory woodland/hardwood forest	Tallgrass	Tallgrass
E23	Bourbon	40c Wooded Osage Plains	Tallgrass/oak hickory woodland/hardwood forest	Tallgrass	Tallgrass

APPENDIX C (continued)

<b>Plot Code</b>	<b>County</b>	<b>EPA Ecoregion IV</b>	<b>EPA prairie designation</b>	<b>NRCS rare and declining habitat</b>	<b>Final designation</b>
E24	Bourbon	40c Wooded Osage Plains	Tallgrass/oak hickory woodland/hardwood forest	Tallgrass	Tallgrass
E32	Bourbon	40c Wooded Osage Plains	Tallgrass/oak hickory woodland/hardwood forest	Tallgrass	Tallgrass
E33	Bourbon	40c Wooded Osage Plains	Tallgrass/oak hickory woodland/hardwood forest	Tallgrass	Tallgrass
E01	Coffey	40b Osage Cuestas	Tallgrass/oak hickory woodland	Tallgrass	Tallgrass
E03	Coffey	40b Osage Cuestas	Tallgrass/oak hickory woodland	Tallgrass	Tallgrass
E07	Coffey	40b Osage Cuestas	Tallgrass/oak hickory woodland	Tallgrass	Tallgrass
E08	Coffey	40b Osage Cuestas	Tallgrass/oak hickory woodland	Tallgrass	Tallgrass
E09	Coffey	40b Osage Cuestas	Tallgrass/oak hickory woodland	Tallgrass	Tallgrass
E15	Coffey	40b Osage Cuestas	Tallgrass/oak hickory woodland	Tallgrass	Tallgrass
E25	Coffey	40b Osage Cuestas	Tallgrass/oak hickory woodland	Tallgrass	Tallgrass
E16	Dickinson	28a Flint Hills	Tallgrass	Tallgrass	Tallgrass
C01	Ellsworth	27a Smoky Hills	Tallgrass/mixed	Tallgrass	Tallgrass
E04	Geary	28a Flint Hills	Tallgrass	Tallgrass	Tallgrass
E13	Greenwood	28a Flint Hills	Tallgrass/oak hickory woodland	Tallgrass	Tallgrass
E14	Greenwood	28a Flint Hills	Tallgrass/oak hickory woodland	Tallgrass	Tallgrass

APPENDIX C (continued)

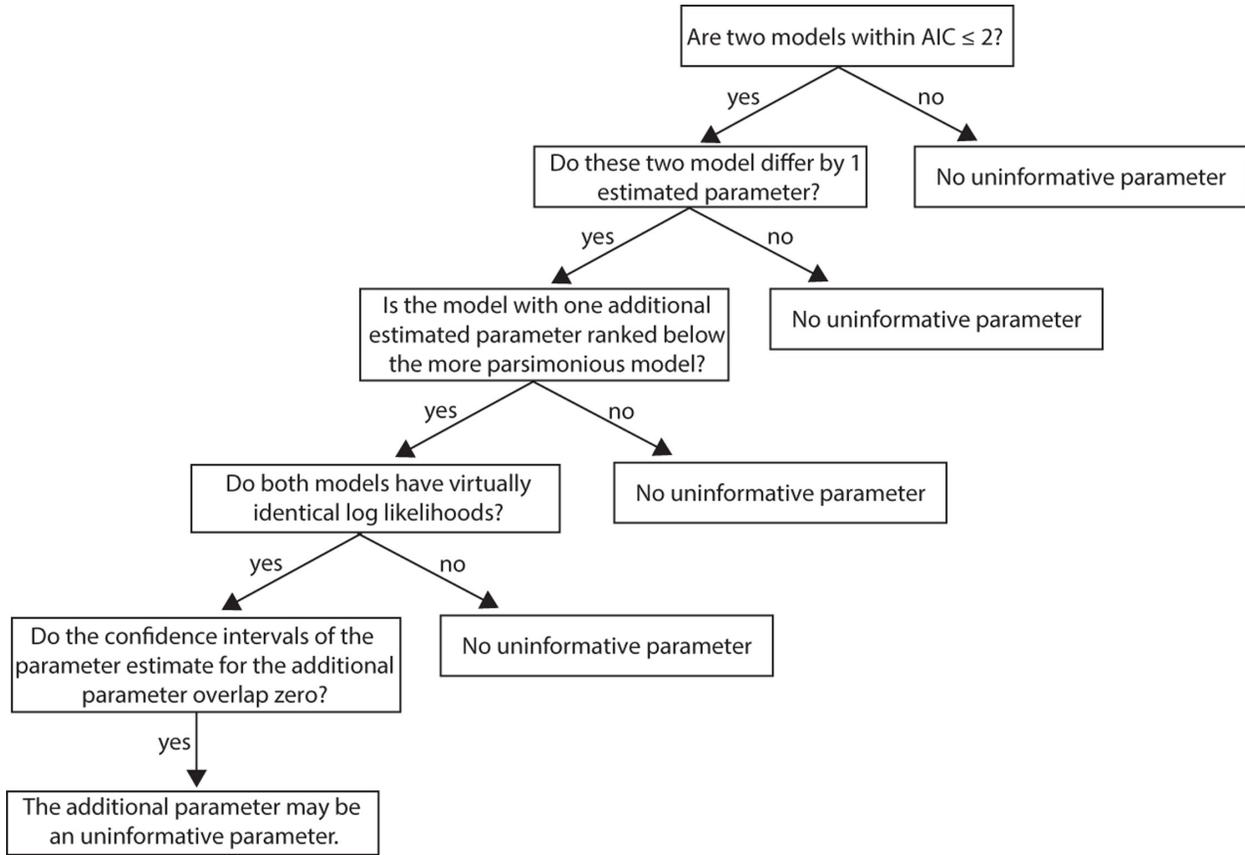
<b>Plot Code</b>	<b>County</b>	<b>EPA Ecoregion IV</b>	<b>EPA prairie designation</b>	<b>NRCS rare and declining habitat</b>	<b>Final designation</b>
E18	Linn	40c Wooded Osage Plains	Tallgrass/oak hickory woodland/hardwood forest	Tallgrass	Tallgrass
E19	Linn	40c Wooded Osage Plains	Tallgrass/oak hickory woodland/hardwood forest	Tallgrass	Tallgrass
E20	Linn	40c Wooded Osage Plains	Tallgrass/oak hickory woodland/hardwood forest	Tallgrass	Tallgrass
E21	Linn	40c Wooded Osage Plains	Tallgrass/oak hickory woodland/hardwood forest	Tallgrass	Tallgrass
E11	Lyon	28a Flint Hills	Tallgrass	Tallgrass	Tallgrass
E34	Lyon	28a Flint Hills	Tallgrass	Tallgrass	Tallgrass
E35	Lyon	28a Flint Hills	Tallgrass	Tallgrass	Tallgrass
E36	Lyon	28a Flint Hills	Tallgrass	Tallgrass	Tallgrass
E17	Marion	28a Flint Hills	Tallgrass	Tallgrass	Tallgrass
E06	Morris	28a Flint Hills	Tallgrass	Tallgrass	Tallgrass
E02	Osage	40b Osage Cuestas	Tallgrass/oak hickory woodland	Tallgrass	Tallgrass
E12	Osage	40b Osage Cuestas	Tallgrass/oak hickory woodland	Tallgrass	Tallgrass
C02	Rice	27a Smoky Hills	Tallgrass/mixed	Tallgrass/Sand	Tallgrass
C03	Rice	27a Smoky Hills	Tallgrass/mixed	Tallgrass/Sand	Tallgrass

APPENDIX C (continued)

<b>Plot Code</b>	<b>County</b>	<b>EPA Ecoregion IV</b>	<b>EPA prairie designation</b>	<b>NRCS rare and declining habitat</b>	<b>Final designation</b>
C15	Rice	27a Smoky Hills	Tallgrass/mixed	Tallgrass/Sand	Tallgrass
C16	Rice	27a Smoky Hills	Tallgrass/mixed	Tallgrass/Sand	Tallgrass
C17	Rice	27a Smoky Hills	Tallgrass/mixed	Tallgrass/Sand	Tallgrass
E05	Wabaunsee	28a Flint Hills	Tallgrass	Tallgrass	Tallgrass
E10	Wabaunsee	40b Osage Cuestas	Tallgrass/oak hickory woodland	Tallgrass	Tallgrass
E26	Woodson	40b Osage Cuestas	Tallgrass/oak hickory woodland	Tallgrass	Tallgrass
E27	Woodson	40b Osage Cuestas	Tallgrass/oak hickory woodland	Tallgrass	Tallgrass

APPENDIX D

LEROUX (2019) UNINFORMATIVE PARAMETER DECISION-MAKING TREE



APPENDIX E

NATIVE BEE ABUNDANCE BY GENERA AND YEAR

Genus (40 total)	2018	2019	Total
<i>Agapostemon</i>	132	670	802
<i>Andrena</i>	15	36	51
<i>Anthidium</i>	1	1	2
<i>Anthophora</i>	7	1	8
<i>Apis</i>	25	123	148
<i>Ashmeadiella</i>	1	0	1
<i>Augochlorella</i>	1186	291	1477
<i>Augochloropsis</i>	28	30	58
<i>Bombus</i>	175	136	311
<i>Calliopsis</i>	4	24	28
<i>Ceratina</i>	11	9	20
<i>Coelioxys</i>	4	4	8
<i>Colletes</i>	4	20	24
<i>Diadasia</i>	27	11	38
<i>Dianthidium</i>	1	0	1
<i>Dieunomia</i>	2	1	3
<i>Epeolus</i>	0	1	1
<i>Eucera</i>	2	0	2
<i>Florilegus</i>	0	7	7
<i>Halictus</i>	199	315	514
<i>Heriades</i>	2	0	2
<i>Holcopasites</i>	0	4	4
<i>Hoplitis</i>	3	13	16
<i>Hylaeus</i>	82	41	123
<i>Lasioglossum</i>	1892	4627	6519
<i>Megachile</i>	52	41	93
<i>Melissodes</i>	133	71	204
<i>Nomada</i>	0	17	17
<i>Nomia</i>	8	4	12
<i>Perdita</i>	47	22	69
<i>Protandrena</i>	29	46	75
<i>Pseudopanurgus</i>	18	10	28
<i>Sphecodes</i>	13	24	37
<i>Stelis</i>	0	4	4
<i>Svastra</i>	61	10	71
<i>Tetraloniella</i>	13	16	29
<i>Triepeolus</i>	15	12	27
<i>Xenoglossa</i>	2	0	2
<i>Xeromelecta</i>	0	1	1
<i>Xylocopa</i>	105	28	133
<b>Grand Total</b>	<b>4,299</b>	<b>6,271</b>	<b>10,970</b>

APPENDIX F

PARAMETER TESTS FOR MULTICOLLINEARITY OF UNIVARIATE PREDICTORS

Year Bee measure	Predictor 1	Predictor 2	Test	Test statistic	P
2018 Abundance and Richness	Floral cover	Floral richness	Spearman's correlation	Sp. coef =0.41	
		CP	Wilcoxon rank-sum	W=1274	0.28
		Grazing	Wilcoxon rank-sum	W=1543	0.60
	Floral richness	Ecoregion	Kruskal-Wallis rank-sum	X-sq.= 9.50	0.02*
		CP	Wilcoxon rank-sum	W=1369.5	0.87
		Grazing	Wilcoxon rank-sum	W=1480	0.89
	Ecoregion	Kruskal-Wallis rank-sum	X-sq.= 30.71	<<0.05*	
2018 Diversity	Floral cover	Floral richness	Spearman's correlation	Sp. coef=0.41	
		CP	Wilcoxon rank-sum	W=1261	0.41
		Grazing	Wilcoxon rank-sum	W=1500	0.55
	Floral richness	Ecoregion	Kruskal-Wallis rank-sum	X-sq.= 8.40	0.04*
		CP	Wilcoxon rank-sum	W=1217.5	0.27
		Grazing	Wilcoxon rank-sum	W=1433	0.86
	Ecoregion	Kruskal-Wallis rank-sum	X-sq.= 31.33	<<0.05*	
2019 Abundance/ Richness	Floral cover	Floral richness	Spearman's correlation	Sp. coef=0.32	
		CP	Wilcoxon rank-sum	W=1398.5	0.94
		Grazing	Wilcoxon rank-sum	W=1926.5	0.002*
	Floral richness	Ecoregion	Kruskal-Wallis rank-sum	X-sq=8.41	0.04*
		CP	Wilcoxon rank-sum	W=1398.5	0.87
		Grazing	Wilcoxon rank-sum	W=1849.5	0.01*
	Ecoregion	ANOVA	X-sq.= 13.94	0.003*	
2019 Diversity	Floral cover	Floral richness	Spearman's correlation	Sp. coef= 0.29	
		CP	Wilcoxon rank-sum	W-1308	0.77
		Grazing	Wilcoxon rank-sum	W=1699	0.004*
	Floral richness	Ecoregion	Kruskal-Wallis rank-sum	X-sq.= 9.53	0.02*
		CP	Wilcoxon rank-sum	W=1269.5	0.98
		Grazing	Wilcoxon rank-sum	W=1632.5	0.01*
	Ecoregion	ANOVA	X-sq.=35.79	<<0.05*	

APPENDIX G

MULTICOLLINEAR PARAMETER COMPARISONS OF FIT FOR UNIVARIATE METRICS

Year	Bee response	Predictor	df	logLik	AICc	delta	weight
2018	Bee abundance	*Floral cover	3	-157.315	320.9	0	0.999
		Ecoregion	5	-162.153	334.9	14.03	0.001
		*Ecoregion	3	-164.963	336.2	1.26	0.347
		*Floral richness	5	-162.153	334.9	0	0.653
	Bee richness	*Floral cover	3	-248.527	503.3	0	1
		Ecoregion	5	-257.213	525	21.73	0
		*Floral richness	3	-255.301	516.8	0	0.984
		Ecoregion	5	-257.213	525	8.18	0.016
	Bee diversity	*Floral cover	3	-176.388	359	0	0.993
		Ecoregion	5	-179.155	368.9	9.9	0.007
		*Floral richness	3	-177.769	361.8	0	0.973
		Ecoregion	5	-179.155	368.9	7.14	0.027
2019	Bee abundance	*Floral cover	3	-198.581	403.4	0	0.796
		Grazing	3	-199.943	406.1	2.72	0.204
		*Grazing	3	-199.943	406.1	0	0.971
		Floral richness	3	-203.462	413.2	7.04	0.029
		*Ecoregion	5	-169.992	350.6	0	1
		Floral cover	3	-198.581	403.4	52.82	0
		*Ecoregion	5	-169.992	350.6	0	1
		Floral richness	3	-203.462	413.2	62.58	0
2019	Bee richness	*Floral cover	3	-263.327	532.9	0	0.994
		Grazing	3	-268.51	543.3	10.37	0.006
		*Floral richness	3	-268.36	543	0	0.537
		*Grazing	3	-268.51	543.3	0.3	0.463
		*Ecoregion	5	-258.751	528.1	0	0.916
		Floral cover	3	-263.327	532.9	4.79	0.084
		*Ecoregion	5	-258.751	528.1	0	0.999
		Floral richness	3	-268.36	543	14.86	0.001
2019	Bee diversity	*Floral cover	3	-160.104	326.5	0	0.997
		Grazing	3	-165.781	337.8	11.35	0.003
		*Floral richness	3	-162.13	330.5	0	0.975
		Grazing	3	-165.781	337.8	7.3	0.025
		*Floral cover	3	-160.104	326.5	0	0.942
		Ecoregion	5	-160.7	332	5.58	0.058
		*Floral richness	3	-162.13	330.5	0	0.682
		Ecoregion	5	-160.7	332	1.52	0.318

## APPENDIX H

### PRELIMINARY MODEL SET MODEL PARAMETERS

\*Due to collinearity (2019), floral cover/richness and grazing/ecoregion did not co-occur in the same models. Colons (:) represent interactions between two predictor variables.

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**Floral resources**

Floral cover

Floral richness

Floral cover : Floral richness

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**Treatments**

CP

Ecoregion

Grazing

CP : Ecoregion

CP : Grazing

Ecoregion : Grazing

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**Ecoregion replaced with Floral cover**

CP

Floral cover

Grazing

CP : Floral cover

CP : Grazing

Floral cover : Grazing\*

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**Ecoregion replaced with Floral richness**

CP

Floral richness

Grazing

CP:Floral richness

CP:Grazing

Floral richness : Grazing\*

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**Ecoregion replaced with Floral cover**

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**Grazing replaced with Floral richness**

CP

Floral richness

Floral cover

CP : Floral richness

CP : Floral cover

Floral richness : Floral cover

APPENDIX I

COMPLETE MODEL-AVERAGED PARAMETER ESTIMATES FOR TOP MODEL SETS

<b>BEE ABUNDANCE 2018</b>					
<b>Parameter</b>	<b>Estimate</b>	<b>7.5% C.I.</b>	<b>92.5% C.I.</b>	<b>Full SE</b>	
(Intercept)	2.418	2.194	2.643	0.155	
CP25 v. CP2	0.592	0.307	0.876	0.196	
Floral cover	0.125	0.088	0.162	0.026	
<b>BEE RICHNESS 2018</b>					
<b>Parameter</b>	<b>Estimate</b>	<b>7.5% C.I.</b>	<b>92.5% C.I.</b>	<b>Full SE</b>	
(Intercept)	1.426	1.286	1.567	0.097	
Floral cover	0.056	0.037	0.075	0.013	
CP25 v. CP2	0.143	-0.028	0.313	0.118	
Ungrazed v. grazed	0.101	-0.099	0.301	0.138	
Ungrazed v. grazed: Floral cover	-0.019	-0.053	0.016	0.024	
CP25: Floral cover	0.012	-0.27	0.294	0.195	
CP2: Floral cover	-0.012	-0.294	0.27	0.195	
<b>BEE DIVERSITY 2018</b>					
<b>Parameter</b>	<b>Estimate</b>	<b>7.5% C.I.</b>	<b>92.5% C.I.</b>	<b>Full SE</b>	
(Intercept)	2.726	2.309	3.143	0.288	
Floral cover	0.129	0.021	0.237	0.075	
Ungrazed v. Grazed	0.437	-0.005	0.88	0.305	
CP25 v. CP2	0.308	-0.217	0.833	0.362	
CP25: Floral cover	-0.142	-0.285	0.001	0.098	
Floral richness	0.114	0.026	0.202	0.061	
Ungrazed v. Grazing (CP25)	-0.337	-1.128	0.454	0.545	
CP2	-0.308	-0.833	0.217	0.362	
CP2: Floral cover	0.142	-0.001	0.285	0.098	
CP2 v. CP25 (Ungrazed)	0.337	-0.454	1.128	0.545	

APPENDIX I (continued)

<b>BEE ABUNDANCE 2019</b>					
<b>Parameter</b>	<b>Estimate</b>	<b>7.5% C.I.</b>	<b>92.5% C.I.</b>	<b>Full SE</b>	
(Intercept)	4.141	3.546	4.735	0.3	
Ungrazed v. Grazed	-0.721	-1.165	-0.276	0.224	
Sand v. Mixed	-2.094	-2.882	-1.305	0.398	
Short v. Mixed	0.528	-0.16	1.217	0.347	
Tall v. Mixed	-1.671	-2.333	-1.009	0.334	
Short v. Sand	2.622	1.932	3.312	0.348	
Tall v. Sand	0.423	-0.24	1.086	0.334	
Tall v. Short	-2.199	-2.74	-1.658	0.273	
<b>BEE RICHNESS 2019</b>					
<b>Parameter</b>	<b>Estimate</b>	<b>Full SE</b>	<b>7.5% C.I.</b>	<b>92.5% C.I.</b>	
(Intercept)	1.518	1.215	1.822	0.21	
Ungrazed v. Grazed	0.383	-0.059	0.824	0.305	
Sand v. Mixed	-0.204	-0.795	0.388	0.409	
Short v. Mixed	0.364	-0.014	0.742	0.261	
Tall v. Mixed	-0.056	-0.455	0.343	0.275	
Sand v. Mixed (Ungrazed)	-1.406	-2.078	-0.733	0.463	
Short v. Mixed (Ungrazed)	-0.562	-1.062	-0.062	0.345	
Tall v. Mixed (Ungrazed)	-0.699	-1.201	-0.197	0.346	
CP25 v. CP2	0.306	0.055	0.557	0.173	
CP25 v. CP2 (Ungrazed)	-0.403	-0.76	-0.045	0.247	
Short v. Sand	0.568	0.117	1.02	0.312	
Tall v. Sand	0.148	-0.282	0.577	0.297	
Short v. Sand (Ungrazed)	0.844	0.239	1.449	0.417	
Tall v Sand (Ungrazed)	0.707	0.1	1.313	0.418	
Tall v. Short	-0.421	-0.695	-0.146	0.189	
Tall v. Short (Ungrazed)	-0.137	-0.545	0.271	0.281	
CP2 v. CP25	-0.306	-0.557	-0.055	0.173	
CP2 v. CP25 (Ungrazed)	0.403	0.045	0.76	0.247	

APPENDIX I (cont.)

<b>BEE DIVERSITY 2019</b>				
<b>Parameter</b>	<b>Estimate</b>	<b>Full SE</b>	<b>7.5% C.I.</b>	<b>92.5% C.I.</b>
(Intercept)	2.262	1.976	2.548	0.197
CP2 v. Floral cover	0.056	0.007	0.106	0.034
CP25 v. Floral cover	0.142	0.074	0.21	0.047
Floral cover	0.11	0.063	0.158	0.033
Sand v. Mixed	0.662	-0.175	1.499	0.576
Short v. Mixed	0.172	-0.421	0.765	0.408
Tall v. Mixed	1.591	0.854	2.328	0.508
Ungrazed v. Grazed	1.719	0.69	2.748	0.709
Sand v. Mixed (Ungrazed)	-2.369	-3.719	-1.019	0.93
Short v. Mixed (Ungrazed)	-1.364	-2.544	-0.184	0.813
Tall v. Mixed (Ungrazed)	-2.336	-3.608	-1.064	0.876
Short v. Sand	-0.49	-1.284	0.303	0.547
Tall v. Sand	0.929	0.023	1.836	0.624
Short v. Sand (Ungrazed)	1.005	-0.042	2.052	0.721
Tall v Sand (Ungrazed)	0.033	-1.117	1.183	0.792
Tall v. Short	1.42	0.732	2.108	0.474
Tall v. Short (Ungrazed)	-0.972	-1.916	-0.028	0.651

APPENDIX J

TOP MODEL SETS FOR 2018 AND 2019,  $\Delta AIC_c < 4$

\*\*Abun= bee abundance, Rich= bee richness, Div= bee diversity (effective genera number)  
 \*\*CP= restoration type, Flw cov= floral cover, Flw rich= floral richness, \*w= model AIC weight,  
 ER= evidence ratio

Year	Bee response***	Model rank	CP**	Grazing	Ecoregion	Flw cover**	Flw rich**	CP : Grazing	CP : Ecoregion	Grazing : Ecoregion	Flw cover : Flw rich	CP : Flw cover	CP : Flw rich	Grazing : Flw cov	df	logLik	AICc	$\Delta AIC_c$ or $\Delta QAIC_c$			
																		w*	ER*		
2018	Abun	1	■			■									4	-152.83	314.00	0	0.97		
	Rich	1				■									3	-248.53	428.50	0	0.23	1.00	
		2	■			■							■		5	-246.2	428.56	0.06	0.23	1.09	
		3	■			■									4	-247.48	428.73	0.23	0.21	1.91	
		4		■		■									4	-248.28	430.08	1.58	0.11	1.10	
		5	■			■									5	-247.22	430.29	1.78	0.1	2.00	
		6		■		■								■	5	-248.1	431.77	3.27	0.05	1.00	
		7	■			■									■	6	-246.92	431.78	3.28	0.05	1.25
		8	■			■			■							6	-247.22	432.28	3.78	0.04	
	Div	1				■									3	-176.39	359.01	0	0.49	2.88	
		2	■			■							■		6	-174.14	361.13	2.12	0.17	1.13	
		3	■			■		■							4	-176.48	361.35	2.34	0.15	1.25	
		4				■		■							3	-177.77	361.77	2.76	0.12	1.71	
		5	■			■		■		■			■		7	-173.93	362.99	3.98	0.07		
	2019	Abun	1			■									6	-164.83	342.5	0	0.98		
Rich $\Delta QAIC$		1			■					■				9	-253.29	459.88	0	0.54	2.08		
		2			■									5	-258.75	461.37	1.49	0.26	2.36		
		3	■		■			■						8	-256.32	463.15	3.27	0.11	1.22		
		4	■		■			■						7	-257.65	463.45	3.57	0.09			
Div		1											■	4	-158.98	326.38	0	0.47	1.04		
		2					■							3	-160.1	326.46	0.07	0.45	5.63		
	3			■	■								9	-154.92	329.81	3.43	0.08				

APPENDIX K

MULTIVARIATE DISPERSION AND PERMANOVA RESULTS

Dispersion\*

\*Dispersion estimates are for non-transformed data. All singletons were removed from the data set prior to analyses.

Year	Factor and levels	F	df	P	
2018, 2019	Year	5.816	1,189	0.027	
2018	CP (CP-2, CP-25)	1.248	1,100	0.328	
	Grazing (ungrazed, grazed)	3.243	1,100	0.114	
	Ecoregion	2.053	3,98	0.197	
	<b>t</b>				
	Tallgrass, sand	0.746		0.495	
	Tallgrass, mixed	2.268		0.056	
	Tallgrass, shortgrass	1.812		0.112	
	Sand, mixed	1.431		0.177	
	Sand, shortgrass	0.702		0.544	
	Mixed, shortgrass	0.38		0.743	
2019		<b>F</b>	<b>df</b>		
	CP (CP-2, CP-25)	0.229	1,87	0.692	
	Grazing (ungrazed, grazed)	0.1523	1,87	0.728	
	Ecoregion	3.846	3,85	0.046	
	<b>t</b>				
	Tallgrass, sand	2.593		0.043	
	Tallgrass, mixed	2.388		0.032	
	Tallgrass, shortgrass	3.046		0.03	
	Sand, mixed	0.487		0.691	
	Sand, shortgrass	0.261		0.832	
Mixed, shortgrass	0.295		0.811		

APPENDIX K (continued)

PERMANOVA results\*

\***Bolded type** for factors indicates marginal or significant differences, asterisks (\*) indicate significant p-values (P<0.05), periods ( . ) indicate marginally significant p-values (P<0.09)

Year	Factor	df	SS	MS	Pseudo-F	P
2018	CP	1,89	3572.3	3572.3	1.465	0.139
	Grazing	1,89	2270.9	2270.9	0.931	0.472
	<b>Ecoregion</b>	3,89	35174	11725	4.809	0.001*
	CP:Grazing	1,89	2120.6	3391.4	0.87	0.532
	<b>CP:Ecoregion</b>	3,89	10174	2117.1	1.391	0.096 .
	Grazing:Ecoregion	3,89	6351.3	2438.1	0.868	0.63
	Residuals	89	2.1699 E5	2438.1		
	Total	101	2.834 E5			
2019	CP	1,76	2393.8	2393.8	0.096	0.487
	Grazing	1,76	2150.3	2150.3	0.865	0.554
	<b>Ecoregion</b>	3,76	64505	21502	8.65	0.001 *
	CP:Grazing	1,76	2697.8	2697.8	1.085	0.337
	CP:Ecoregion	3,76	7035	2345	0.0943	0.548
	Grazing:Ecoregion	3,76	7690.5	2563.5	1.031	0.407
	Residuals	76	1.8892 E5	2485.8		
	Total	88	2.7927 E5			

APPENDIX K (continued)

*Post-hoc* pairwise comparison\*

\***Bolded type** for factors indicates significant differences, asterisks (\*) indicate significant p-values (p<0.05)

Year	Factor	Groups	Den. Df	t	P
2018	Ecoregion	<b>Tallgrass, sand</b>	53	1.909	0.003*
		<b>Tallgrass, mixed</b>	57	2.04	0.004*
		<b>Tallgrass, shortgrass</b>	70	3.096	0.001*
		<b>Sand, mixed</b>	28	1.777	0.009*
		<b>Sand, shortgrass</b>	41	2.315	0.001*
		Mixed, shortgrass	45	1.207	0.172
	CP-25	<b>Tallgrass, sand</b>	28	2.046	0.003*
		<b>Tallgrass, mixed</b>	30	1.628	0.021*
		<b>Tallgrass, shortgrass</b>	36	2.809	0.001*
		<b>Sand, mixed</b>	14	1.757	0.013*
		<b>Sand, shortgrass</b>	20	3.075	0.001*
		Mixed, shortgrass	22	1.264	0.17
	CP-2	Tallgrass, sand	19	1.018	0.424
		<b>Tallgrass, mixed</b>	21	1.402	0.049*
		<b>Tallgrass, shortgrass</b>	28	1.938	0.001*
		Sand, mixed	8	1.019	0.415
		Sand, shortgrass	15	1.029	0.391
		Mixed, shortgrass	17	1.087	0.288
2019	Ecoregion	<b>Tallgrass, sand</b>	39	1.651	0.005*
		<b>Tallgrass, mixed</b>	47	3.154	0.001*
		<b>Tallgrass, shortgrass</b>	63	4.536	0.001*
		<b>Sand, mixed</b>	22	1.797	0.013*
		<b>Sand, shortgrass</b>	38	2.41	0.001*
		Mixed, shortgrass	46	1.216	0.159

APPENDIX L

INDICATOR GENERA BY TREATMENT GROUP

\*Indicative groups and genera within groups across both years are designated with an asterisk. Only significant indicator genera are reported. \*\*Sqrt I.V. = the square-root of the indicator value and computed in with statistical software R v.3.5.2 (R Core Team 2018) using the “indicspecies” package (De Caceres and Legendre 2009)

Group	Level	2019			2018		
		Indicator genus	Sqrt I.V.**	P	Indicator genus	Sqrt I.V.**	P
CP	2	<i>Hylaeus</i>	0.196	0.047	<i>Diadasia</i>	0.21	0.022
		<i>Calliopsis</i>	0.136	0.033			
	25	No significant genera			<i>Lasioglossum</i>	0.274	0.003
					<i>Pseudopanurgus</i>	0.148	0.042
Grazing	Ungrazed	<i>Svastra</i>	0.305	0.003	No significant genera		
	Grazed	<i>Lasioglossum</i>	0.235	0.027			
Ecoregion*	Shortgrass	<i>Melissodes</i>	0.406	0.008	<i>Anthophora</i>	0.431	0.001
		* <i>Triepeolus</i>	0.402	0.011	* <i>Triepeolus</i>	0.274	0.027
		<i>Agapostemon</i>	0.335	0.019	<i>Diadasia</i>	0.285	0.023
					<i>Andrena</i>	0.274	0.047
	Sand	<i>Xylocopa</i>	0.399	0.002	<i>Apis</i>	0.269	0.038
	Mixed	* <i>Pseudopanurgus</i>	0.34	0.012	* <i>Pseudopanurgus</i>	0.274	0.013
		<i>Hoplitis</i>	0.314	0.021			
	Tallgrass	<i>Apis</i>	0.411	0.004	<i>Augochlorella</i>	0.379	0.005
		* <i>Hylaeus</i>	0.359	0.023	* <i>Hylaeus</i>	0.337	0.01
					<i>Bombus</i>	0.267	0.046
	Mixed /Short	* <i>Lasioglossum</i>	0.435	0.009	* <i>Lasioglossum</i>	0.301	0.034
		<i>Svastra</i>	0.302	0.041			
	Sand/Short	No significant genera			<i>Perdita</i>	0.322	0.018

# APPENDIX M

## NMDS VISUALIZATIONS OF 2018 BEE COMMUNITIES BY CP AND ECOREGION

